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Final Technical Report

**“Restoring Sustainable Forests on Appalachian Mined Lands for Wood Products,
Renewable Energy, Carbon Sequestration, and Other Ecosystem Services”**

Final Report

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ABSTRACT

Concentrations of CO₂ in the Earth's atmosphere have increased dramatically in the past 100 years due to deforestation, land use change, and fossil fuel combustion. These human-caused, higher levels of CO₂ may enhance the atmospheric greenhouse effect and may contribute to climate change. Many reclaimed coal-surface mine areas in the eastern U.S. are not in productive use. Reforestation of these lands could provide societal benefits, including sequestration of atmospheric carbon. The goal of this project was to determine the biological and economic feasibility of restoring high-quality forests on the tens of thousands of hectares of mined land and to measure carbon sequestration and wood production benefits that would be achieved from large-scale application of forest restoration procedures.

We developed a mine soil quality model that can be used to estimate the suitability of selected mined sites for carbon sequestration projects. Across the mine soil quality gradient, we tested survival and growth performance of three species assemblages under three levels of silvicultural. Hardwood species survived well in WV and VA, and survived better than the other species used in OH, while white pine had the poorest survival of all species at all sites. Survival was particularly good for the site-specific hardwoods planted at each site. Weed control plus tillage may be the optimum treatment for hardwoods and white pine, as any increased growth resulting from fertilization may not offset the decreased survival that accompanied fertilization.

Grassland to forest conversion costs may be a major contributor to the lack of reforestation of previously reclaimed mine lands in the Appalachian coal-mining region. Otherwise profitable forestry opportunities may be precluded by these conversion costs, which for many combinations of factors (site class, forest type, timber prices, regeneration intensity, and interest rate) result in negative land expectation values. Improved technology and/or knowledge of reforestation practices in these situations may provide opportunities to reduce the costs of converting many of these sites as research continues into these practices.

It also appears that in many cases substantial payments, non-revenue values, or carbon values are required to reach "profitability" under the present circumstances. It is unclear when, or in what form, markets will develop to support any of these add-on values to supplement commercial forestry revenues. However, as these markets do develop, they will only enhance the viability of forestry on reclaimed mined lands, although as we demonstrate in our analysis of carbon payments, the form of the revenue source may itself influence management, potentially mitigating some of the benefits of reforestation.

For a representative mined-land resource base, reforestation of mined lands with mixed pine-hardwood species would result in an average estimated C accumulation in forms that can be harvested for use as wood products or are likely to remain in the soil C pool at ~250 Mg C ha⁻¹ over a 60 year period following reforestation. The "additionality" of this potential C sequestration was estimated considering data in scientific literature that defines C accumulation in mined-land grasslands over the long term. Given assumptions detailed in the text, these lands have the potential to sequester ~180 Mg C ha⁻¹, a total of 53.5 x 10⁶ Mg C, over 60 years, an average of ~900,000 Mg C / yr, an amount equivalent to about 0.04% of projected US C emissions at the midpoint of a 60-year period (circa 2040) following assumed reforestation. Although potential sequestration quantities are not great relative to potential national needs should an energy-related C emissions offset requirement be developed at some future date, these lands are available and unused for other economically valued purposes and many possess soil

and site properties that are well-suited to reforestation. Should such reforestation occur, it would also produce ancillary benefits by providing environmental services, such as enhanced watershed protection, and producing timber and renewable-fuel products.

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INTRODUCTION

Problem Statement

Concentrations of CO₂ in the earth's atmosphere have increased dramatically in the past 100 years due to deforestation, land use change, and fossil fuel combustion. These human-caused, higher levels of CO₂ may enhance the atmospheric greenhouse effect and may contribute to climate change. Agencies and institutions within the U.S. government are exploring ways to reduce greenhouse gas emissions and sequester CO₂ while maintaining energy production and economic prosperity. Minimizing emissions caused by deforestation and sequestering CO₂ via reforestation of previously forested lands may significantly reduce atmospheric CO₂ and help achieve the goal of reducing risks associated with climate change.

To achieve carbon sequestration goals, it is necessary to develop a broad suite of options that have potential to sequester significant amounts of carbon, are effective over long time periods, and do not introduce new environmental problems. Sequestering carbon in high-value forests on lands mined for coal is an option that meets these criteria. Furthermore, most coal-mined land in the East was covered with forests prior to mining. The forest is a large carbon sink that is removed in the process of mining coal. The coal is used for energy production with consequent CO₂ emissions. Therefore, it is logical to restore mined lands to productive forest systems that will maximize carbon recapture in vegetation, litter and soil.

According to U.S. Department of Agriculture estimates (1979), approximately 4 million hectares of land were mined for coal prior to passage of the federal Surface Mining Control and Reclamation Act (SMCRA) in 1977. Since that time, millions of additional hectares have been affected by coal mining. For example, over 1.8 million hectares nationally (including 1.1 million hectares in the east) were under active coal mining permits during 2001; of these lands, over 600,000 hectares are currently classified as "disturbed" (OSM, 2002). Although an accurate measure of the total acreage converted from forests to grassland since the implementation of the SMCRA is not available, we believe it is between 150,000 and 170,000 hectares in the eastern U.S. alone.

Public Law 95-87, the Surface Mining Control and Reclamation Act (SMCRA), mandates that mined land be reclaimed in a fashion that renders the land at least as productive after mining as it was before (Torbert et al. 1995). Research has shown that restored forests on mined lands can be equally as or more productive than the native forests removed by mining (Burger and Zipper 2002). Given that most land surface-mined for coal in the Appalachians was originally forested, forestry is a logical land use for most of the reclaimed mined land in the region (Torbert and Burger 1990). However, since implementation of the SMCRA, fewer forests are being restored in the eastern and midwestern coalfield regions (Burger et al. 1998). In several states, most notably Virginia, the majority of mined land is being restored to "unmanaged forests," but these restored forests are generally not productive due to low site quality and inadequate species selection. Region-wide, the majority of mined land that was originally forested is not being reclaimed in a way that favors tree establishment, timber production, carbon sequestration, and long-term forest productivity (Torbert and Burger 1990).

Reclaimed mined lands are producing timber and sequestering carbon at rates far below their potential for reasons that include poor mine soil quality, inadequate stocking of trees, lack of reforestation incentives, and regulatory disincentives for planting trees on previously forested land (Boyce 1999, Burger and Maxey 1998). A number of these problems can be ameliorated

simply through intensive silvicultural management. Through established site preparation techniques such as ripping, weed control, fertilizing, and liming, the quality of a given site can be improved. Other management and silvicultural techniques such as site-species matching, correct planting techniques, employing optimal planting densities, post-planting weed control, and thinning can also improve normal development of forest stands and improve timber production and carbon sequestration.

Similar to the much-debated topic of converting agricultural land to forests, the conversion of reclaimed mined lands to forests carries with it many economic implications. The primary difference between converting agricultural lands to forests and converting reclaimed mined lands to forests is the absence of any obvious extrinsic opportunity cost in the latter scenario; this, of course, assumes that the reclaimed mined land has been abandoned and is not being utilized for any economically beneficial purpose. There is a lack of research pertaining specifically to the conversion of reclaimed mined lands from their current uses to forests and the economic implications of such a land use conversion. Furthermore, the potential for an incentive scheme aimed at promoting the conversion of reclaimed mined lands to forests has yet to be explored in depth.

This project addressed DOE Area of Interest 4, Terrestrial Carbon Sequestration. The specific DOE research objective is: *“Demonstrate and verify large-scale carbon sequestration by reforestation of post-mining lands using high-value tree species.”* Our overall goals and objectives were to evaluate the biological and economic feasibility of restoring high-quality forests on the tens of thousands of hectares of mined land and to measure carbon sequestration and wood production benefits that would be achieved from large-scale application of forest restoration procedures. The Intergovernmental Panel on Climate Change uses the following criteria for judging carbon sequestration program/project success: sequestration potential, technology, cost, environmental impact, public acceptance, and legal issues. This project addressed these criteria within five project tasks led by six investigators.

1. Assess carbon sequestration and timber production potentials on mine soils of varying site quality (J. A. Burger).
2. Develop field investigation procedures for use in assessing site characteristics that determine reforestation and carbon sequestration potentials, and a system for applying those procedures to classify such lands according to reforestation site quality (J. Galbraith).
3. Develop methods and procedures for reforesting mined lands of varying site qualities so as to enhance carbon sequestration potentials (T. R. Fox).
4. Evaluate economic returns to landowners that would be likely to occur through application of specific reforestation and forest management techniques intended to enhance carbon sequestration on post-SMCRA grasslands of varying site qualities, and evaluate the potential for policy incentives to encourage landowner adoption of such practices (G. Amacher, J. Sullivan).
5. Determine the potential of large-scale post-SMCRA grassland restoration to sequester carbon and create other societal benefits (C.E. Zipper).

EXECUTIVE SUMMARY

Concentrations of CO₂ in the Earth's atmosphere have increased dramatically in the past 100 years due to deforestation, land use change, and fossil fuel combustion. These human-caused, higher levels of CO₂ may enhance the atmospheric greenhouse effect and may contribute to climate change. Agencies and institutions within the U. S. government are exploring ways to reduce greenhouse gas emissions and sequester CO₂ while maintaining energy production and economic prosperity. Minimizing emissions caused by deforestation and sequestering CO₂ via reforestation of previously forested lands may significantly reduce atmospheric CO₂ and help achieve the goal of reducing risks associated with climate change.

Many reclaimed coal-surface mine areas in the eastern US are not in productive use. Reforestation of these lands could provide societal benefits, including environmental services such as watershed protection, sequestration of atmospheric carbon (C), and production of wood products. Currently, little information is available on the characteristics and extent of the land resource base created under SMCRA, and little is known about the potential for eastern US coal-mined land to sequester carbon were these lands successfully reforested

This project addressed DOE Area of Interest 4, Terrestrial Carbon Sequestration. The specific DOE research objective is: *“Demonstrate and verify large-scale carbon sequestration by reforestation of post-mining lands using high-value tree species.”* Our overall goals and objectives were to evaluate the biological and economic feasibility of restoring high-quality forests on the tens of thousands of hectares of mined land and to measure carbon sequestration and wood production benefits that would be achieved from large-scale application of forest restoration procedures. The Intergovernmental Panel on Climate Change uses sequestration potential, technology, cost, environmental impact, public acceptance, and legal issues as criteria for judging carbon sequestration program/project success. This project addressed these criteria within five project tasks led by multiple investigators.

1. Assess carbon sequestration and timber production potentials on mine soils of varying site quality.
2. Develop field investigation procedures for use in assessing site characteristics that determine reforestation and carbon sequestration potentials, and a system for applying those procedures to classify such lands according to reforestation site quality.
3. Develop methods and procedures for reforesting mined lands of varying site qualities so as to enhance carbon sequestration potentials.
4. Evaluate economic returns to landowners that would be likely to occur through application of specific of reforestation and forest management techniques intended to enhance carbon sequestration on post-SMCRA grasslands of varying site qualities, and evaluate the potential for policy incentives to encourage landowner adoption of such practices.
5. Determine the potential of large-scale post-SMCRA grassland restoration to sequester carbon and create other societal benefits.

The work was based on several experimental sites located throughout the Appalachian and Midwestern Coalfields, including 14 afforested mined sites varying in age from 20 to 56 years located in a seven-state area of the eastern coalfields; a 26-yr-old white pine stand on mined land in Virginia; a newly-reclaimed mine site planted to oaks that were 3 years old when studied; and a new field study located in Ohio, West Virginia, and Virginia that included three forest types

(white pine, hybrid poplar, mixed hardwood) and three silvicultural regimes (competition control, competition control plus tillage, competition control plus tillage plus fertilization).

Task 1: We developed a thermal oxidation technique for differentiating sequestered soil carbon from inorganic and fossilized carbon found at high levels in mine soils. The methodology provided a means to correct for the carbon loss from the more volatile constituents of coal fragments in the mine soil samples, and another correction factor for the protected organic matter that could remain un-oxidized following thermal pretreatment at high oxidation temperatures (~900°C).

We developed a geospatial and statistical protocol for carbon monitoring and accounting which included the horizontal and vertical distributions and variability of sequestered soil organic carbon (SOC), and the maximum cost-effective depth (MCD) and the minimum detectable difference (MDD) of SOC. Regression equations were used to describe terrestrial carbon sequestration in total tree biomass, leaf litter, and soils for pine, mixed, and hardwood forests established on mined land in the Midwestern and Appalachian coal fields.

On average, the highest amount of ecosystem carbon on mined land was sequestered by pine stands (148 Mg ha⁻¹; at a rate of 4.19 Mg C ha⁻¹ yr⁻¹), followed by hardwood stands (130 Mg ha⁻¹; at a rate of 3.01 Mg C ha⁻¹ yr⁻¹) and mixed (pine and hardwood) stands (118 Mg ha⁻¹; at a rate of 3.12 Mg C ha⁻¹ yr⁻¹). Non-mined hardwood stands sequestered approximately 42%, 62%, and 79% more cumulative carbon in total tree biomass, litter, and soils, than the mined pine, mined hardwood, and mined mixed stands, respectively. For the mined forest stands, approximately three-quarters of the total ecosystem carbon was sequestered in the total tree biomass (stem wood, stem bark, foliage, treetops, branches, stumps, and coarse roots) and the remaining one-quarter was distributed between the litter and soil carbon pools. As expected, site quality had an exponential effect on ecosystem carbon sequestration in both pine and mixed stands, while ecosystem carbon decreased with stand age due to natural pine tree deterioration occurring in older pine stands and in the pine component of mixed stands.

Finally, with Task 1, we developed the framework of a decision support system (DSS) consisting of the first iteration of a dynamic model to predict carbon sequestration for a 60-year period for three forest types (white pine, hybrid poplar, and native hardwoods) at three levels of management intensity: low (weed control), medium (weed control and tillage) and high (weed control, tillage, and fertilization). On average, the highest amount of ecosystem carbon on mined land was sequestered by pine stands (148 Mg ha⁻¹), followed by hardwood (130 Mg ha⁻¹) and mixed stands (118 Mg ha⁻¹). Non-mined hardwood stands contained 210 Mg C ha⁻¹, which was about 62% higher than the average of all mined stands. After 60 years, the net carbon in ecosystem components, wood products and landfills ranged from 20 to 235 Mg ha⁻¹ among all scenarios. The highest net amount of carbon was estimated under mixed hardwood vegetation established by the highest intensity treatment. Under this best case scenario, a surface-mined land of average site quality would sequester net carbon stock at 235 Mg C ha⁻¹, at a rate of 3.9 Mg C ha⁻¹ yr⁻¹, which was 100% greater than a grassland scenario.

Task 2: Successful establishment of a productive forest on reclaimed mined land can provide economic benefits through wood production, wildlife habitat, watershed protection, and carbon sequestration. The SMCRA of 1977 requires that reclaimed land be equally as, or more

productive than pre-mined conditions. However, since the passage of this law, few productive forests have been established, largely due to the inability to estimate mine soil quality for forests.

We developed a forest site quality productivity model for mine soils that incorporated key soil properties most influenced forest growth on mine soils:

$$PI = (pH \times \text{texture} \times \text{density} \times CF)^{1/4} \times \text{depth}$$

where PI = Productivity Index; pH = sufficiency of pH; texture = sufficiency of texture; density = sufficiency of soil density; CF = sufficiency of coarse fragments; and depth = sufficiency of rooting depth. A PI was calculated for each of 52 white pine sites located in VA, WV and KY using the measured values for pH, texture, density, CF, and depth. PI values were regressed with white pine SI to determine the extent to which this general PI model correlated with SI. The fit of the general PI model to SI of the validation sites resulted in an R^2 value of 0.63. This shows that this general soil-based PI model could be used to estimate the site quality of reclaimed mined land and, and make judgments about the suitability of selected mined sites for carbon sequestration projects.

Task 3: Forest practices for mined land have not been developed for post-law mined sites. We tested survival and growth performance of three species assemblages under three levels of silvicultural input intended to alleviate the adverse conditions for tree survival and growth on post-SMCRA mined land. The test included three forest types (white pine, hybrid poplar, mixed hardwood) and three silvicultural regimes (competition control, competition control plus tillage, competition control plus tillage plus fertilization). White pine and hardwood species grew little over the course of the first growing season, which could translate into a need for continued weed control to ensure the trees do not succumb to the competing vegetation as mean heights ranged from 25 to 40 cm for hardwoods and from 20 to 30 cm for white pine. Hardwood species survived well in WV and VA, and survived better than the other species used in OH, while white pine had the poorest survival of all species at all sites. Survival was particularly good for the site-specific hardwoods planted at each site. Weed control plus tillage may be the optimum treatment for hardwoods and white pine, as any increased growth resulting from fertilization may not offset the decreased survival that accompanied fertilization. Hybrid poplar appeared to have good potential for reverting post-SMCRA mined lands reclaimed to grasses back to forests, as this species had good growth with 50 to 65 cm of height growth in one year in WC+T at all sites and excellent growth in WC+T+F in VA. This good growth, coupled with survival percentages that may be adequate to ensure that without further weed control, an adequately stocked stand could develop, gives this species an advantage over the other species used in this study.

Task 4: We examined the financial viability of reforesting reclaimed surface mined lands in the Appalachian coal-mining region, addressed the economic implications of a shift in reforestation burden from the landowner to the mine operator or government, and explored the opportunities for forest carbon payments on these lands. Our analysis is conducted by comparing land expectation value (LEV) for a range of land use conversion scenarios in which the landowner assumes the costs of site preparation and tree planting, versus scenarios in which the coal company assumes the costs. The comparisons are then used to determine the level of carbon payment required to make forest land-use a financially viable option in circumstances where it is not already viable.

Grassland to forest conversion costs may be a major contributor to the lack of reforestation of previously reclaimed mine lands in the Appalachian coal-mining region. Otherwise profitable forestry opportunities may be precluded by these conversion costs, which for many combinations of factors (site class, forest type, timber prices, regeneration intensity, and interest rate) result in negative LEVs. Improved technology and/or knowledge of reforestation practices in these situations may provide opportunities to reduce the costs of converting many of these sites as research continues into these practices.

It also appears that in many cases substantial payments, non-revenue values, or carbon values are required to reach “profitability” under the present circumstances. It is unclear when, or in what form, markets will develop to support any of these add-on values to supplement commercial forestry revenues. However, as these markets do develop, they will only enhance the viability of forestry on reclaimed mined lands, although as we demonstrate in our analysis of carbon payments, the form of the revenue source may itself influence management, potentially mitigating some of the benefits of reforestation.

Task 5: In order to estimate the potential of lands mined for coal under SMCRA to sequester atmospheric C, if those lands were to be successfully reforested, those lands’ total acreages were estimated from US Office of Surface Mining data, assuming 50% of the total of Phase III bond released lands (1978-2005) and 50% of Phase I bond-released lands over the 2001-2005 period in the study region (eastern OH, PA, eastern KY, WV, VA, MD, TN) (a total of about 300,000 hectares) to be available for reforestation. The total potential of these lands to accumulate atmospheric C was estimated, as a function of SI50, using relationships developed by Amichev (2007) through analysis of data collected by Rodrigue (2001). Assuming the sampled mine sites to be representative of the mined-land resource base, reforestation of mined lands with mixed pine - hardwood species would result in an average estimated C accumulation in forms that can be harvested for use as wood products or are likely to remain in the soil C pool at $\sim 250 \text{ Mg C ha}^{-1}$ over the 60-year period following reforestation.

Only when C accumulates as a response to management actions in quantities over and above that which would accumulate in the absence of such management can that C be considered as “sequestered” a condition commonly termed as “additionality.” The “additionality” of this potential C sequestration was estimated considering data in scientific literature that defines C accumulation in mined-land grasslands over the long term. Given assumptions detailed in the text, these lands have the potential to sequester $\sim 180 \text{ Mg C ha}^{-1}$, a total of $53.5 \times 10^6 \text{ Mg C}$, over 60 years, an average of $\sim 900,000 \text{ Mg C / yr}$, an amount equivalent to about 0.04% of projected US C emissions at the mid-point of a 60-year period (circa 2040) following assumed reforestation.

Although potential sequestration quantities are not great relative to potential national needs should an energy-related C-emissions offset requirement be developed at some future date, these lands are available and unused for other economically valued purposes and many possess soil and site properties that are well-suited to reforestation. Should such reforestation occur, it would also produce ancillary benefits by providing environmental services, such as enhanced watershed protection, and producing timber and renewable-fuel products. As time passes, these lands’ collective economic potential for reforestation can be expected to diminish due to the influence of continued proliferation and growth of low-value woody species with minimal C sequestration potentials.

TASK 1: Estimate forest productivity and carbon sequestration potential on mined lands supporting abandoned grasslands.

Executive Summary

The goal of Task 1 was to determine the extent to which reforestation of mined lands would sequester atmospheric CO₂ and how these carbon sequestration rates compared to the current rates of sequestration under abandoned grassland conditions, typical of post-mining reclamation. The work within this task consisted of five sub-studies: (1) develop a soil organic carbon measurement procedure for mine soils; (2) develop a geospatial and statistical basis for mine soil sampling for carbon sequestration accounting; (3) establish a protocol for carbon monitoring and accounting on mined land; (4) develop a carbon sequestration empirical model for forests and soils on mined land in the Midwestern and Appalachian coalfields; and (5) develop a decision support system based on expert knowledge for predicting carbon sequestration for forest establishment practices on mined land.

Sub-study 1 successfully differentiated SOC from coal-derived carbon for mine soils in the hard rock region of the Appalachian coalfield. The methodology provided a means to correct for the carbon loss from the more volatile constituents of coal fragments in the mine soil samples, and another correction factor for the protected organic matter that could remain unoxidized following thermal pretreatment at high oxidation temperatures (~900°C). The SOC estimates were between 15 and 23% accurate for mine soils derived from sandstone and siltstone spoils, which are commonly used for reclamation in the hard rock region of the Appalachian coalfield.

Sub-study 2 helped resolve carbon inventory and analysis issues for mined land. The objective of this work was to determine the horizontal and vertical distributions and variability of sequestered soil organic carbon (SOC), and the maximum cost-effective depth (MCD) and the minimum detectable difference (MDD) of SOC. Soil samples from topsoil and spoil materials from shallow (to 30 cm depth) and deep (to 200 cm depth) soil pits were collected from mined lands reclaimed to grasslands located in Ohio (OH), Virginia (VA), and West Virginia (WV). The volume of excavated mine soil samples was transformed in terms of costs of sampling and carbon analysis, and the measured SOC content was transformed in terms of profits from carbon credits in order to determine the MCD for carbon inventorying on these sites. Based on the variation of SOC (vertically and horizontally), we determined the MDD of SOC between two consecutive carbon inventories. The SOC content for the surface 0-10 cm layer in OH (1,115 g C m⁻²) was significantly higher compared to the sites in WV (427 g C m⁻²) and VA (148 g C m⁻²). Due to the presence of coal particles in the samples, the coal-derived carbon ranged from 118 to 964% greater than plant-derived carbon among all sites. In one growing season, very little carbon was added to mine soils. The MCD analysis for the OH2 site indicated that the profit break-even depth for carbon inventory was 18 cm at sampling intensity of 17 plots ha⁻¹. Due to the lower horizontal variability of the OH1 and OH3 sites, the MCD on these sites exceeded 200 cm. The overall findings from this project supported the understanding that establishing carbon sequestration projects on mined lands could be a profitable undertaking for mined land reclamation.

Sub-study 3 produced a mined land-specific field sampling protocol for ecosystem carbon analysis that would enable mined land owners to measure and report sequestered carbon in the form of vegetation biomass, leaf litter, and SOC. Organized in a four-step approach, the procedures in the protocol aimed at: (step A) establishing permanent carbon monitoring plots

across a mined site; (step B) baseline carbon stock estimation; (step C) periodic carbon content measurement; and (step D) reference areas. The purpose of the protocol was to provide consistent and practical guidelines for estimation of ecosystem carbon content of mined lands at various stages of floristic succession, ranging from abandoned grasslands to shrubland and forest land, so that these estimates would meet the accuracy and precision of a future global market. Due to high compaction and high rock content in mine soils, some of the procedures in this protocol aimed at estimation of the minimum detectable difference (MDD) and maximum cost-effective depth (MCD) for carbon analysis in mine soils. Both MDD and MCD are critically important tools that would reduce the costs for carbon analysis and reporting on mined land.

Sub-study 4 produced regression equations to describe terrestrial carbon sequestration in total tree biomass, leaf litter, and soils for pine, mixed, and hardwood forests established on mined land in the Midwestern and Appalachian coalfields. We used data for 14 mined and 8 adjacent, non-mined forests to determine the carbon sequestration potential of mined land reclaimed prior to the passage of SMCRA. On average, the highest amount of ecosystem carbon on mined land was sequestered by pine stands (148 Mg ha^{-1} ; at a rate of $4.19 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$), followed by hardwood stands (130 Mg ha^{-1} ; at a rate of $3.01 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$) and mixed (pine and hardwood) stands (118 Mg ha^{-1} ; at a rate of $3.12 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$). Non-mined hardwood stands sequestered approximately 42%, 62%, and 79% more cumulative carbon in total tree biomass, litter, and soils, than the mined pine, mined hardwood, and mined mixed stands, respectively. For the mined forest stands, approximately three-quarters of the total ecosystem carbon was sequestered in the total tree biomass (stem wood, stem bark, foliage, treetops, branches, stumps, and coarse roots) and the remaining one-quarter was distributed between the litter and soil carbon pools. As expected, site quality had an exponential effect on ecosystem carbon sequestration in both pine and mixed stands, while ecosystem carbon decreased with stand age due to natural pine tree deterioration occurring in older pine stands and in the pine component of mixed stands.

Sub-study 5 used the above regression equations as building components for a first iteration of a decision support system (DSS) designed to predict the amount of sequestered carbon on mined lands of different forest type and different forest establishment treatments. The DSS was designed to track the amount of sequestered carbon with time in each ecosystem component (tree biomass, leaf litter, soil) and harvested wood biomass (wood products in-use and in-landfills). The objective was to develop the structure of an interactive tool that could be used to evaluate the effectiveness of reforestation for carbon sequestration on reclaimed mined land suitable for use by private landowners, regional planners, and politicians.

The overall findings from this study showed that forest establishment on mined land could sequester significant amounts of atmospheric CO_2 . The net carbon sequestration measured at the end of a 60-year period (one hardwood stand rotation) ranged from 20 to 235 Mg C ha^{-1} for forest establishment scenarios based on three species groups (white pine, hybrid poplar, and mixed hardwoods) and three intensities of silvicultural treatments (weed control (WC), WC plus soil tillage (T), and WC+T plus fertilization (F)). The highest net amount of sequestered carbon was estimated under mixed hardwood vegetation established by the highest intensity treatment, WC+T+F. Under this scenario, a mined land of average site quality ($\text{SI} = 25 \text{ m}$) would sequester net carbon stock at 235 Mg C ha^{-1} (sequestered at a rate of $3.9 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$) at the end of the first 60-year rotation. About 71% of the net sequestered carbon was in above-ground biomass, followed by the carbon stock in root biomass (14%), soil (11%), and litter layer biomass (4%).

If reforestation were selected for surface-mined land reclamation, there could be significant implications for the role of mined lands on the amount of sequestered carbon in these terrestrial ecosystems. If all surface-mined land located in the eastern regions of the United States (approximately 650,000 ha [OSM 2007]) was planted to mixed hardwood forests, the cumulative net carbon sequestration from forest activities on these lands (estimated at 2.5×10^6 Mg C yr⁻¹) could account for nearly 3% of the Kyoto emission reduction requirement for the United States (7% below 1990 emissions).

Task 1, Sub-study 1: Soil organic carbon measurement procedure for mine soils

Experimental

The purpose of this study was to develop a laboratory method for organic carbon estimation in mine soils based on the work of Schmidt et al. (2001) and Kuhlbusch (1995) related to black carbon quantification in soils. The method is a combination of a chemical pretreatment for carbonate removal, a thermal oxidation pretreatment to remove all volatile organic carbon forms currently present in the mine soil samples, followed by an analysis of elemental carbon mass difference by an automated carbon analyzer (Vario MAX CNS analyzer, elemental, Hanau, Germany). The objectives of this research were to: (1) develop a new technique for measuring carbon content on reclaimed mined land that differentiates between pedogenic carbon and geogenic carbon present in mine soils; (2) evaluate and compare the performance (accuracy and precision) of the new carbon measurement method against the Walkley-Black procedure for carbon analysis in mine soils; and (3) determine the effects of spoil material type and coal content in mine soils on the accuracy and precision of the Walkley-Black procedure.

Mine Soils and Standard Materials

Mine soil material was obtained from an active mine near the town of Norton, Virginia. The materials were adjacent to the Taggart and Taggart Marker coal seams of the Marcum Hollow member of the Upper Wise Formation. We collected weathered SS and SiS fragments and coal fragments from the 0-50 cm mine soil profile of a reclaimed mined site that is typical for the hard-rock region of the Appalachian coalfield (Daniels and Amos 1982) (Fig. 1-1-1).



Figure 1-1-1. Weathered sandstone and siltstone rock fragments and coal fragments obtained from a reclaimed mined site near the town of Norton, Virginia. The photographs show the three materials prior to grinding with porcelain mortar and pestle.

All fragments were washed with tap water, after which they were dried at 60°C for 30-60 minutes to remove all residual surface moisture prior to grinding. Washing of all rock fragments was done to assure that no interference was caused by any organic particles adsorbed to the surfaces or lodged in the macropores.

A very fine coating of silt- and clay-sized soil particles was removed from the surfaces of nearly all rock and coal fragments, evident from the color difference of these fragments before and after washing. Because SOM and microscopic coal fragments are commonly associated with the silt and clay fractions of the soil, washing the rock and coal fragments was assumed to be sufficient to assure that the soil mixtures constructed for this study from ground coal and weathered SS and SiS rock fragments were free of contamination; that is, no microscopic coal particles were associated with the collected weathered SS and SiS rock fragments, and no SOM or soil minerals were associated with the collected coal fragments (Fig. 1-1-1). Because all constructed soils were treated for carbonates prior to elemental carbon analysis, any contamination of the soil mixtures created by soluble carbonates introduced with tap water was eliminated.

Following washing and grinding, all fragments were passed through sieve No. 60 to assure consistency of SOM, mineral particles, and coal fragment sizes ($< 250\ \mu\text{m}$) in all 30 mine soil mixtures that were constructed for the experiment. Following sieving, precise amounts of coal were mixed in with the ground-weathered SS and SiS materials in order to create the standard mine soils. The ground coal and the weathered SS/SiS in each constructed mine soil were thoroughly mixed in glass jars of approximately 500 ml volume. The entire soil volume in the glass jars was shaken for 1 hr at an approximate shaking speed of 120-150 cycles per minute. Therefore, the resulting mine soil mixtures consisted of a quantitatively known amount of homogeneously mixed coal material.

In addition to the weathered SS and SiS fragments, we used soil samples from the A-horizon of a Jefferson forest soil series (fine-loamy, siliceous, mesic Typic Hapludults), as well as industrial grade quartz sand to construct two additional soils. We used quartz sand to represent a mine soil of extremely low, nearly zero carbon content such as those formed from mineral sand mining. In contrast, the Jefferson forest soil was chosen to represent a natural, unlimed topsoil of higher SOC content compared to the SS and SiS materials. All constructed standard mine soils were used to test the precision and accuracy of SOC estimates by the proposed new carbon partitioning technique and by the Walkley-Black method.

Detailed Procedure for Determining SOC in Mine Soils

The following is a detailed description of our proposed 16-step method for measuring SOC in mine soils designed for this project and tested on a total of 30 different soil mixtures representing mine soils in the hard-rock region of the Appalachian coalfield:

1. Collect mine soil samples from a desired depth or desired soil horizon.
2. Collect coal fragments detected during the soil sample collection process. If additional coal fragments are necessary, then scout the site and collect additional coal fragments located at the surface of the mine site. Wash and dry all fragments. Retain for further analysis in Step 5.

3. Separate the coarse rock fragments from the fine earth fraction by sieving the mine soil samples through a sieve No. 10 (< 2 mm). Wash and dry all separated rock fragments. After recording their weight, retain for further analysis in Step 5.
4. Grind approximately 15 g per mine soil sample of the fine soil material (< 2 mm) using mortar and pestle until all ground material of the 15-g subsample is passed through a sieve No. 60 (< 250 μ m).
5. Crush and grind approximately 15 g of the site-specific, soil-free coal fragments and 15 g of the soil rock fragments (from Steps 2 and 3) until all ground materials are passed through sieve No. 60 (250 μ m).
6. From the three ground materials (from Steps 4 and 5), place two replications of approximately 600 mg of the ground soil sample, two replications of approximately 100 mg of the ground coal, and two replications of 600 mg of the ground soil rock fragments in ceramic crucibles that can resist heating up to 1,000°C. Record the crucible ID/label and the corresponding sample weight to the nearest 0.0001 g. Use a separate subsample of the ground soil sample, ground coal, and ground soil rock fragments (about 5 g) to determine their respective moisture contents, where:

$$\text{Moisture, \%} = \frac{[\text{air-dry material, g}] - (\text{oven-dry}_{105^{\circ}\text{C}} \text{ material, g})}{(\text{oven-dry}_{105^{\circ}\text{C}} \text{ material, g}) * 100}$$

7. Add approximately 1 ml of distilled water to each 600-mg ground mine soil sample, as well as to the reference 100-mg ground coal and ground soil rock materials to increase the efficiency of carbonate removal by HCl acid fumigation (Harris et al., 2001).
8. Place between 20 and 30 samples in a vacuum desiccator (5-l volume) along with a 150-ml beaker with 100 ml of concentrated HCl (12 M) to remove soil carbonates. The desiccator is only used to confine the HCl vapors and no vacuum conditions are necessary for this pretreatment.

Repeat this step for the reference ground coal sample to assure consistency of the parallel analysis.

9. Allow all ground samples to be exposed to HCl vapor for a period of 24 hours.
10. After the HCl exposure, remove crucibles from the desiccator and put them in a drying oven at 60°C for 4-6 hours to remove residual moisture.

At this stage of the procedure, all ground mine soil samples will be carbonate-free. If it is necessary to measure the amount of carbonates present in the samples, then duplicate soil samples should be used for elemental carbon analysis before and after the carbonate removal pretreatment using a carbon auto-analyzer.

11. Use the first sample replications to measure the residual elemental carbon of each carbonate-free soil sample, coal sample, and ground soil rock sample using a carbon auto-analyzer with a preset minimum combustion temperature of 900°C.

For the analysis at this step it is important that the original total sample weight is entered manually into the carbon auto-analyzer machine as opposed to entering the sample weight measured after carbonate removal. Failure to enter the correct initial sample weight for its

corresponding soil sample, placed on the feeding tray of the carbon auto-analyzer, will produce incorrect estimates for soil organic carbon concentration (SOC ,wt%) of the sample.

Note that the carbon results reported by the auto-analyzer machine must be corrected for sample moisture content, as shown in Step 15 below.

12. Place the second sample replications into a muffle furnace, preheated to 375°C, to remove all pedogenic organic matter.

The 375°C temperature was the lowest of three chosen temperature settings, along with 550 and 650°C, which were tested to determine the optimal temperature for thermal pretreatment of mine soil samples at which detectable levels of SOC remained for elemental carbon analysis at Step 15. The detection limit of the carbon auto-analyzer used in this study was 0.01% C by sample weight.

13. Allow samples to be exposed to thermal oxidation in the muffle furnace, preset to the chosen temperature, for a period of 24 hours.
14. Remove samples from muffle furnace and place them in a desiccator to allow cooling.
15. Without transferring to a different crucible, measure the residual elemental carbon of each soil sample, coal sample, and ground soil rock sample using a carbon auto-analyzer with a preset minimum combustion temperature of 900°C.

Similar to the analysis at Step 11, for the analyses at this step, it is important that the original total sample weight is entered manually into the carbon auto-analyzer machine as opposed to entering the sample weight measured after the combined effects of the carbonate removal and thermal pretreatments.

The carbon results reported by the auto-analyzer machine (reported_C%) are corrected for sample moisture content as follows:

$$\text{Corrected C(wt\%)} = \frac{\text{reported\_C\%} * (100 + \text{moisture\%})}{100}$$

16. Soil organic carbon concentration is estimated as follows (this equation is also presented as Eq. 1 later in the text):

$$\text{SOC(wt\%)} = \frac{A * (C_0 - B * C_{375})}{A - B}$$

where A and B are ratios computed by dividing the carbon concentrations of the ground soil rock and ground coal samples with no thermal pretreatment from Step 11 by their respective carbon concentrations after thermal pretreatment from Step 15. These ratios are determined during the parallel oxidation analyses of ground soil rock samples (A) and ground coal (B) as depicted in Figure 1-1-1. C_0 is the carbon concentration (wt%) measured in the mine soil sample with no thermal pretreatment (Step 11). The C_0 estimate is the sum of all pedogenic carbon and GC in the soil sample. C_{375} is the carbon concentration (wt%) measured in the second mine soil sample replication after thermal pretreatment for 24 hours (Steps 12-15). The C_{375} estimate includes the thermally-resistant fractions of the pedogenic carbon and GC in the soil sample.

Laboratory Experiment Design

An experiment of 5 x 6 x 3 factorial design was set up to evaluate the accuracy of the 16-step method for five soil materials, six levels of coal fragment content in mine soils, and three thermal oxidation temperature levels (Step 12 of the method). The five soil materials were: (i) weathered SS; (ii) weathered SiS; (iii) a mixture of approximately 50% weathered SS and 50% percent weathered SiS (50:50 SS:SiS); (iv) industrial grade quartz sand; and (v) a laboratory soil standard which was the A horizon of a Jefferson forest soil.

The first two soil materials were collected from an active mine near Norton, Virginia. The chemical and physical properties of the spoil materials were representative of most mined sites in the hard rock regions in the Appalachians (Daniels and Amos, 1982). The weathered SS and weathered SiS materials were carefully chosen to be completely free of coal fragments.

All soil materials were ground to pass a No. 60 mesh sieve ($< 250 \mu\text{m}$ final particle size) in order to eliminate the effects of variable soil mineral and coal fragment particle size on the efficiency of SOC estimation by the proposed method, as coal fragments exist in mine spoils in a wide range of sizes (Daniels and Amos, 1982). Grinding to a standardized soil particle size of less than $250 \mu\text{m}$ also allowed a more efficient means of removing all carbonates present in the sample (Harris et al., 2001). The site-specific coal fragments were also ground to a final particle size of less than $250 \mu\text{m}$ to produce an accurate correction factor for the carbon loss from the more volatile constituents of these fragments occurring during the parallel carbon analysis of the reference coal sample.

The six levels of coal fragment content were similar to those used by Daniels and Amos (1982) in their experiment to evaluate the effect of coal fragments on SOC in mine soils using the Walkley-Black wet oxidation procedure. These levels were: (i) 0%; (ii) 0.5%; (iii) 1.0%; (iv) 2.5%; (v) 5.0%; and (vi) 10.0% of total soil weight.

A total of 30 mine soil sample mixtures were constructed by adding each of the six levels of ground coal material to each of the five ground soil materials previously mentioned. Following carbonate removal by HCl fumigation (Harris et al. 2001; Hedges and Stern 1984; Yamamuro and Kayanne 1995), the 30 soil mixtures were exposed to three thermal oxidation temperature levels: (i) 375°C (adopted from Schmidt et al. 2001), (ii) 550°C (Bendfeldt et al. 2001; Schulten and Leinweber 1999), and (iii) 650°C (derived from Othman and Shamsuddin 2003; Zoller et al. 1999), for a period of 24 hours to determine the effect of temperature on the efficacy of thermal transformation of pedogenic carbon and the more volatile GC forms to CO_2 in each sample. All SOC estimates for the 30 soil mixtures were replicated 5 times for a total of 450 estimates using the 16-step SOC estimation method.

In a separate experiment with a 5x6 factorial design, each of the 30 soil mixtures was analyzed for their organic carbon content using the Walkley-Black procedure (Nelson and Sommers 1982; Walkley and Black 1934). All SOC estimates using the Walkley-Black procedure were replicated 3 times for a total of 90 estimates.

Data Analysis

For the evaluation of the 16-step SOC estimation method, the dependent variable was the percent relative uncertainty from the mean (PRUM, %), which was the ratio between the absolute difference of the estimated SOC and the sample SOC present in the soil in the numerator, and the sample SOC in the denominator, and multiplied by 100 to convert to percent:

$$\left(\text{Abs} \frac{\text{SOC}\%_{\text{estimated}} - \text{SOC}\%_{\text{present}}}{\text{SOC}\%_{\text{present}}} \right) * 100$$

The $\text{SOC}\%_{\text{present}}$ is the SOC measurement of the soil samples with 0%-coal addition, referred to as coal-free soil samples, and $\text{SOC}\%_{\text{estimated}}$ is the SOC of each constructed mine soil mixture. The three independent variables for this experiment were spoil material type, coal fragment content, and thermal oxidation temperature.

Similarly, for the evaluation of the Walkley-Black procedure, the dependent variable was PRUM(%), which was calculated as percent ratio between the absolute difference of the estimated SOC and sample SOC present in the soil in the numerator, and the sample SOC in the denominator, and multiplied by 100 to convert to percent:

$$\left(\text{Abs} \frac{\text{WB-SOC}\%_{\text{estimated}} - \text{SOC}\%_{\text{present}}}{\text{SOC}\%_{\text{present}}} \right) * 100$$

The $\text{WB-SOC}\%_{\text{estimated}}$ is the SOC of each constructed mine soil mixture estimated by the Walkley-Black method. The two independent variables for the evaluation of the Walkley-Black method were spoil material type and coal fragment content.

Analysis of variance (ANOVA) and means separation analysis were performed for both experiments using standard statistical procedures (PROC GLM, in SAS® statistical software package) (SAS 2007). The Walkley-Black carbon estimates were further analyzed to attempt to determine any existing correlation between these estimates and the SOC present in the mine soil samples across the spectrum of spoil material type and coal contents analyzed. The established correlation was modeled to produce a regression model to correct Walkley-Black carbon estimates reported in previously published work for mined lands, such as the SOC estimates reported by Rodrigue (2001) and Akala and Lal (2001a).

We split our Walkley-Black data set into two subsets, a modeling and a testing subset, which we used to develop and test the regression model for correction of Walkley-Black carbon estimates for mine soils. The general form of the model was $\text{SOC, wt}\%_{\text{present}} = a * (\text{material type}) + b * (\text{WB-SOC, \%}_{\text{estimated}})$. In order to incorporate material type, which is a categorical variable, in the regression model, a substitute “dummy” variable was created in our data set labeled *order* where *order* = 1 for SS, = 2 for SiS, and = 3 for 50:50 SS:SiS soil samples. Additionally, the intercept in the model was forced to 0, as noted in the general form above, because of the presence of the dummy variable *order*, which would become the value of the intercept after substituting 1, 2, or 3 for *order* in the final correction model. The dependent and independent variables for the linear regression modeling task using regression analysis procedures (PROC REG) in SAS® statistical analysis software were as follows:

<u>Dependent</u>	<u>Regressors</u>
SOC, wt% _{present}	WB-SOC, % _{estimated} Material Type

Results and Discussion

Steps 1 and 2: Carbonate Removal

For the evaluation of the effectiveness of the carbonate removal procedure from each of the 30 mine soil mixtures, we estimated the difference of carbon content from two elemental carbon analyses before and after HCl acid fumigation. The two elemental carbon analyses necessary for estimation of the amount of carbonate-derived carbon are denoted as [C1] and [C2] analyses of the proposed 16-step SOC estimation method; the carbonate carbon content was estimated as the difference of C(wt%) values as [C1] – [C2].

The carbonate-derived carbon content was measured in the coal-free soil samples to avoid any interference from the coal (Fig. 1-1-2). On average, the highest amount of carbonate-derived carbon was present in the SiS soil samples (0.38 wt%, percent by sample weight), followed by the 50:50 SS:SiS samples (0.16 wt%), and SS samples (0.05 wt%). The carbonate carbon contents of the quartz sand and Jefferson forest soil materials were negligible. The ground quartz sand was obtained from a commercially available source and was advertised to be near 100 percent silicon dioxide. The Jefferson soil was acquired from the A horizon of a forest site that had never been limed; all carbonates that may have existed in the soil would have dissolved and leached.

The results for the quartz sand with added coal indicated that the total carbon of the sample significantly increased following HCl fumigation and sample drying at 60°C (prior to elemental carbon analysis) implying that carbon was added to these samples. The amount of added carbon in the quartz sand soil samples increased as the relative coal content (weight %) increased in the soil mixtures, especially for the samples with greater than 2.5wt% coal content (Figure 1-1-2).

For the soil mixtures that contained carbonates, the carbonate-derived carbon appeared to decrease as the coal content in the samples increased (Fig. 1-1-2). One possible explanation of this phenomenon could be that CO₂ from air was incorporated in the soil mixtures via a chemical reaction promoted by the presence of coal either during HCl fumigation or during the 4- to 6-hr period of sample drying. While it was not clear as to what mechanisms promoted the addition of air-CO₂ in the soil, there was an observable relationship between coal content present in the soil mixtures and the decreased amount of measured carbonate carbon. An alternative explanation for the above phenomenon could be that due to the presence of coal, the efficiency of the carbonate removal procedure for some of the soil materials could be reduced.

Although there was no observable trend for the SS and Jefferson forest soil materials, in the case of the SiS and the 50:50 SS:SiS soil materials, the presence of coal in the samples appeared to negate the efficiency of the HCl fumigation technique to remove carbonates. For every 1 wt% of coal added to the SiS and the 50:50 SS:SiS soil materials, the efficiency of carbonate removal by HCl fumigation decrease by 6.5 and 4.6%, respectively (Fig. 1-1-2).

The presence of coal in the soil could either inhibit carbonate removal by HCl fumigation, or the coal could promote an unspecified process of carbon addition to the soil sample from an external source, such as air, during the HCl fumigation treatment or during sample drying prior to elemental carbon analysis (Fig. 1-1-2). Another possibility could be that coal particles absorb the CO₂ from the air, based on the fact that a common method of methane extraction from coal mines is by pumping CO₂ gas, which purges the methane gas absorbed to the coal (White et al. 2005).

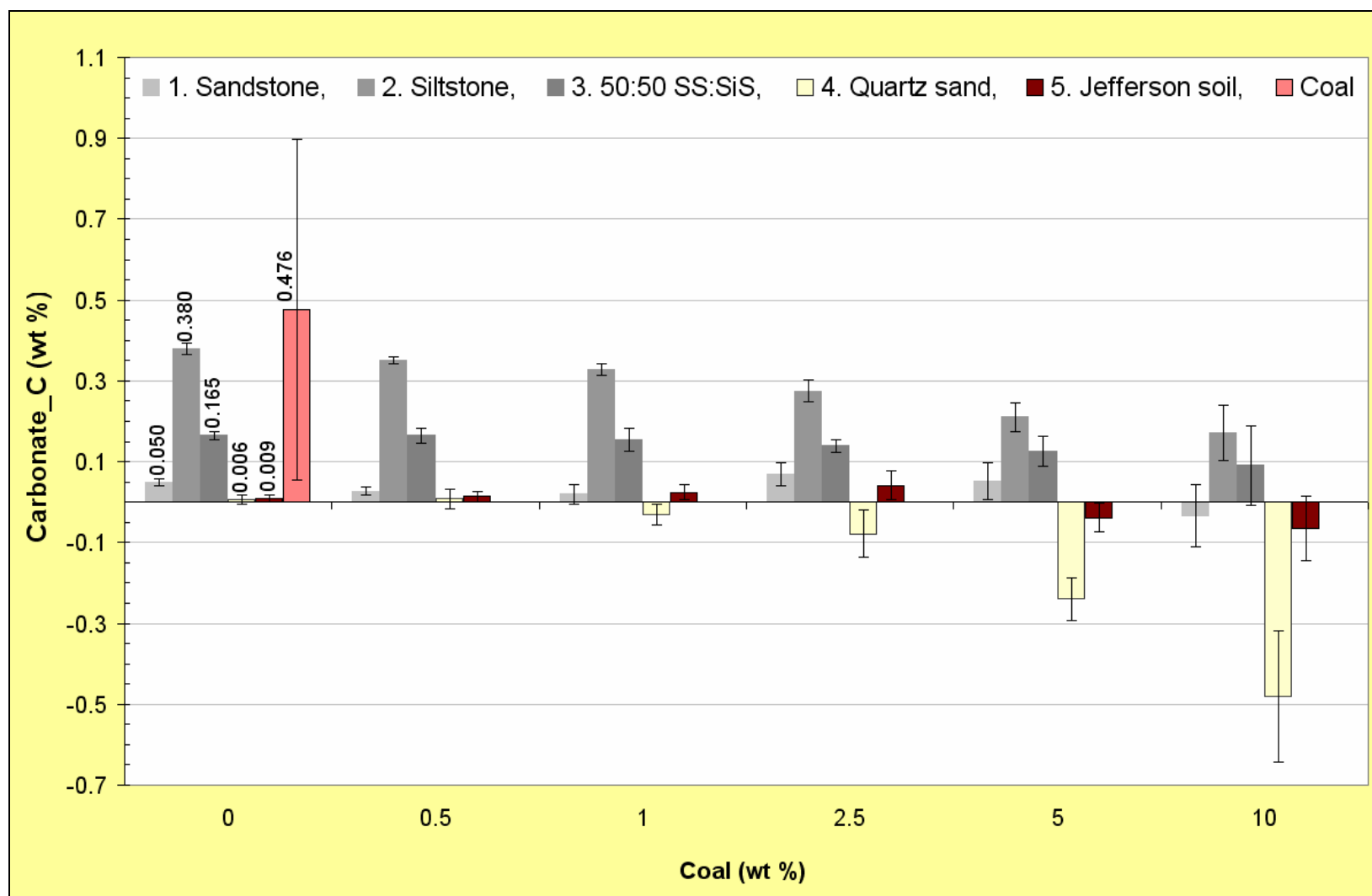


Figure 1-1-2. Carbonate-derived carbon content in mine spoil mixtures of different material type, ranging from SS, SiS, 50:50 SS:SiS mixture, quartz sand, and a laboratory soil standard (Jefferson forest soil) at six levels of coal content expressed as percent by total sample weight. Vertical lines represent one standard error of the mean. Negative carbonate content (quartz sand) implies that carbon addition to soil samples could occur following HCl fumigation and sample-drying, most likely due to CO₂ (from air) absorption to the coal particles in these samples.

The possibility that CO₂ absorption from the air to the coal particles in the samples could be occurring is supported by the data presented in Figure 1-1-2, which showed that more carbon was present in the soil samples of higher coal content, especially for samples of greater-than-2.5wt% coal, following the carbonate removal pretreatment. For quartz sand mixtures, which were carbonate-free, proportionally greater amount of CO₂ seemed to be added (absorbed to the surfaces of coal particles) to the samples as coal content increased (Fig. 1-1-2). Additional work may be needed to determine the exact causes and experimental conditions that may have led to carbon additions (most likely CO₂ absorption). For all SOC analyses in this project, we assumed that all carbonates were successfully removed by HCl fumigation from all soil samples, and that CO₂-absorption was insignificant relative to the SOC concentrations in these samples.

Steps 3 and 4: Pedogenic Carbon Results

The organic matter content in the five material types ranged from 0.03 to 0.81 wt% in the following decreasing order: Jefferson forest soil > SiS > 50:50 SS:SiS > SS > quartz sand (Fig. 1-1-3). The carbon content of the coal fragments used in this study was about 83wt%, which was similar to the estimates obtained from other researchers working on mined lands in the Appalachian coalfields (Li 1991).

The carbon content of the SiS and SS materials (crushed soil rock fragments) contained about 78 and 10%, respectively, of the SOC content in the Jefferson forest soil. The organic carbon content in the SiS material was nearly 670% more than the SS, which supported our previous argument that some spoil materials could contain sufficient amount of old organic carbon within the soil rock structures (Figure 1-1-3). Similar to microbial decomposition of coal (Rumpel and Kogel-Knabner, 2002), as mine soils develop, the old organic matter in the soil rocks could be decomposed and included in the SOC cycle. Because of the dramatic differences in the amount of old organic carbon contained in soil rock fragments of different origin, carbon inventories should be designed to include or account for this important soil carbon pool.

Oven temperature had a significant effect on SOC measurements (Figure 1-1-3). Compared to the no pre-treatment samples, the 375°C temperature treatment oxidized nearly 99.99% of the organic carbon in the SS and quartz sand materials, and about 92, 91, and 94% of the organic carbon in the SiS, 50:50 SS:SiS, and Jefferson forest soil materials, respectively. At this oxidation temperature treatment, about 99.73% of the coal carbon content was oxidized to CO₂. At 375°C, the organic carbon contents for all soil mixtures, except for quartz sand of less than 1wt% coal content, were above the detection limit of the auto-analyzer machine (Fig. 1-1-3).

At 550 and 650°C temperature treatments, the content of the resistant organic carbon in all soil materials was below the detection limit, which deemed these temperatures not practical for the proposed SOC estimation method. Compared to 375°C, as the treatment temperature was increased to 550°C, an additional 88% of the coal carbon was oxidized to CO₂. There was no further oxidation of the coal carbon at 650°C compared to the 550°C temperature, implying that the carbon content in the coal measured after the 550°C temperature treatment, measured at 0.026wt%, was from the most resistant components in the coal material characterized with highly-condensed aromatic molecular structure.

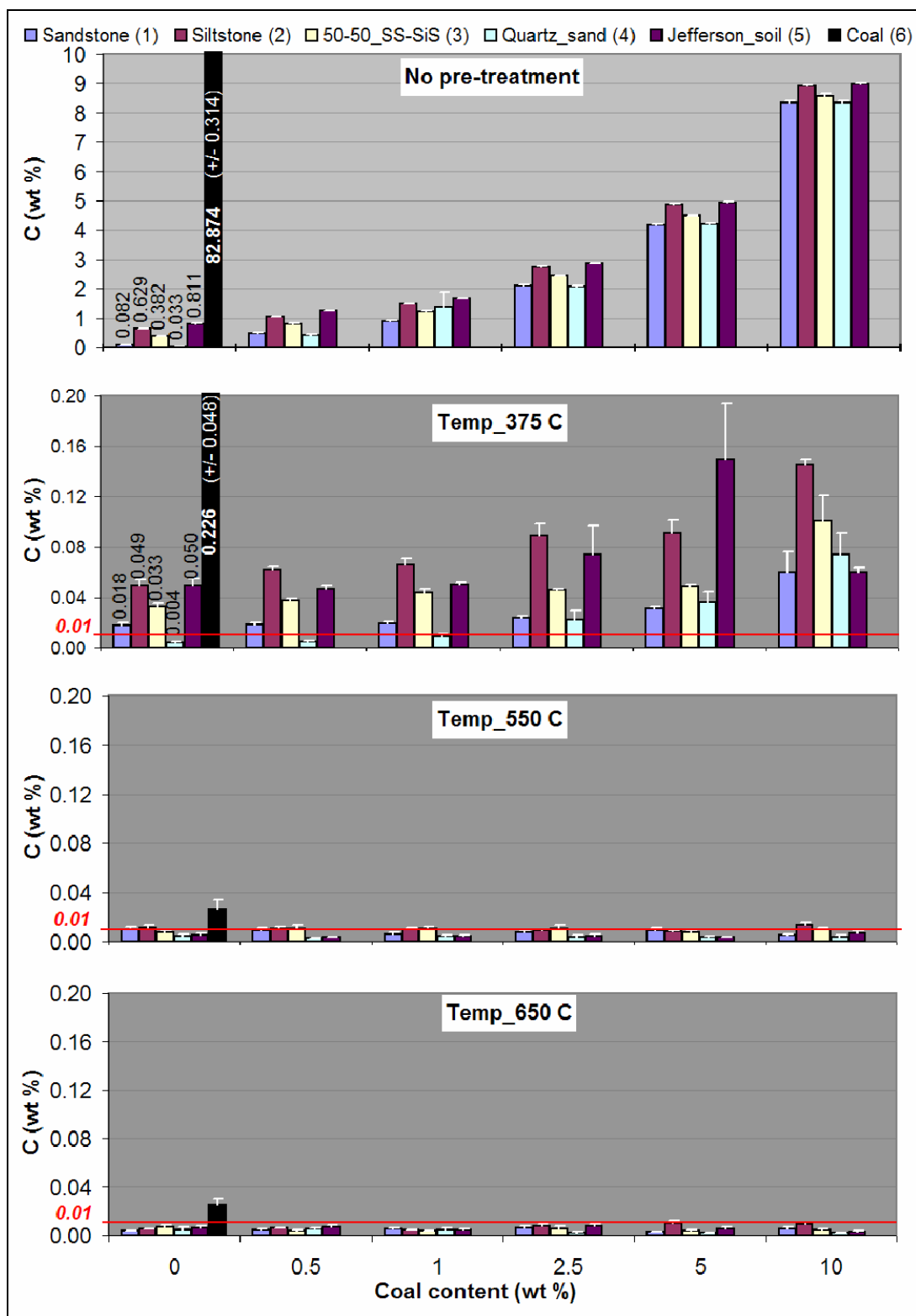


Figure 1-1-3. Average carbon contents (wt%) of 30 carbonate-free spoil mixtures and coal fragments after a 24-hour thermal treatment at three experimental temperatures, 375, 550, and 650°C. Vertical lines represent the standard error of the mean. The horizontal line at 0.01% indicates the detection limit of the carbon auto-analyzer machine used in this study.

Because carbonates are resistant to thermal degradation, they were removed prior to elemental carbon analysis. The results for the five coal-free material types indicated that the amount of carbonates could be significant in certain mine spoil materials (Fig. 1-1-2). The carbonate carbon content in the SS and SiS materials was 38 % of STC; STC is the sum of SIC and SOC. In the coal-free spoil material comprised of approximately 50:50 SS:SiS, the carbonate-derived carbon content was 30 % of STC. This suggests that it is important to remove carbonates from mine soils before SOC analysis, as the failure to do so would result in 40 to 60% overestimation of the SOC present in mine soils formed from SS and SiS overburden spoil material (Figs. 1-1-2 and 1-1-3).

One of the important steps in the proposed SOC estimation procedure was to determine the optimal temperature for thermal treatment of soil samples at which all pedogenic carbon would be released as CO₂ while measurable amounts of the more resistant carbon molecules from coal remained in the sample. Our results indicated that the optimal thermal treatment temperature should be somewhere between 375°C and 550°C (Fig. 1-1-3).

Although the range of thermal temperatures was not sufficient to determine the exact temperature setting for thermal treatment of mine soil samples for SOC analysis, the results show that at 375°C some organic matter molecules remained unoxidized during the 24-hour thermal pretreatment (Fig. 1-1-3). These organic matter compounds are most likely associated with clay soil minerals that protected them from thermal degradation.

Because we were careful to suppress black carbon formation during the laboratory work by placing the samples directly in a 375°C-heated oven, similar to the work by Kuhlbusch (1995) and Schmidt et al. (2001), we believe that the protected carbon in the SS- and SiS-containing soil mixtures is from the organic matter embedded in the SS and SiS rock sediments during deposition, where it was preserved and protected from microbial decomposition for millions of years. Similarly, we believe that all detectable organic carbon following 375°C thermal treatment of the Jefferson forest soil originated from the soil parent material (i.e., from old organic matter of highly aromatic chemical structure) and was protected by the clay- and the silt-soil particles present in this soil (Fig. 1-1-3). For the coal-free soil materials, our results indicated that the temperature-resistant, old organic carbon content of the Jefferson forest soil was 6% of the total SOC in this soil, and was 22% for the SS material, 8% for the SiS material, and 9% for the 50:50 SS:SiS material (Fig. 1-1-3).

The presence of old organic matter in soils would have no effect on the final SOC estimate that would be claimed as carbon credits. The carbon credits for a certain C-sequestration project are estimated as the difference between the SOC content measured at the end of the project and the baseline SOC content for the project area. The baseline SOC content is the soil carbon present in the soil prior to implementation of any project treatments such as tree planting and fertilization. As a result, the SOC from old organic matter, which will also be included in the baseline carbon content, will be subtracted from the SOC measured at the end of the project. This is plausible because the old organic matter found in the structures of rock fragments is expected to be uniformly distributed across the mined site and down the mine soil profile. In the remainder of this text, the term “pedogenic carbon” will be used to describe the amount of carbon derived from organic matter either sequestered in recent times or protected in the structures of soil rock fragments.

In contrast, the distribution of coal-derived carbon in mine soils is expected to follow the random distribution of coal fragments across the mined site and down the mine soil profile. Hence, it is important that the coal-derived carbon be accurately and precisely accounted for in both the baseline and project-end SOC estimates for all mined land carbon sequestration projects.

At temperatures equal to or greater than 550°C, all carbon forms in the soil samples, including the most resistant coal-derived carbon and carbon in the protected organic matter, were oxidized to CO₂ and were released from the samples. Despite the relatively good detection limit (0.01 wt%) of the carbon auto-analyzer used in this project, there was no detectable amount of elemental carbon remaining in the samples for carbon analysis in all 550°C- and 650°C-samples (Fig. 1-1-3). Therefore, the 16-step SOC estimation method for mine soils was developed solely from carbon results for the no pre-treatment and 375°C pre-treatment samples.

The no pre-treatment soil samples included all pedogenic carbon and GC. In order to partition the total carbon in mine soils (after carbonates have been removed) to pedogenic and geogenic C, we solved the following four simultaneous equations:

$$P_0 + G_0 = C_0$$

$$P_{375} + G_{375} = C_{375}$$

$$P_0 / P_{375} = A$$

$$G_0 / G_{375} = B$$

where P_0 = pedogenic carbon concentration (wt%) in the no pre-treatment sample, including recent and protected organic matter; G_0 = geogenic carbon concentration (wt%) in the no pre-treatment sample; P_{375} = temperature-resistant pedogenic carbon concentration (wt%) in the 375°C-sample; G_{375} = temperature-resistant geogenic carbon concentration (wt%) in the 375°C-sample; A and B = ratios computed by dividing the pedogenic carbon and GC concentrations before thermal pre-treatment by the respective carbon concentrations present in the sample after a 375°C thermal treatment. These ratios represent the relative carbon loss from pedogenic (P_0/P_{375}) and geogenic (G_0/G_{375}) carbon compounds present in the soil due to the 24-hour exposure to 375°C; C_0 = carbon concentration (wt%) measured from a soil sample with no pre-treatment, which is the sum of all SOM- and coal- derived carbon in the soil; C_{375} = carbon concentration (wt%) measured after a thermal treatment for 24 hours at 375°C, which includes the temperature-resistant SOM- and coal- derived carbon amounts in the soil.

Solving the above four equations for P_0 and G_0 yields the following two equations, which will be used to determine the amount of pedogenic and geogenic C, respectively, present in all 30 soil mixtures:

$$\text{Pedogenic C (wt\%)} = P_0 = \frac{A * (C_0 - B * C_{375})}{A - B} \quad \text{Eq. 1}$$

$$\text{Geogenic C (wt\%)} = G_0 = \frac{B * (A * C_{375} - C_0)}{A - B} \quad \text{Eq. 2}$$

Because P_0 , P_{375} , G_0 , and G_{375} are the carbon concentrations of distinctive components in a soil sample which can not be physically isolated for measurement, we assumed that the A ratio for coal-free materials (i.e., the 0%-coal soil mixtures) will stay the same when coal is added to these mixtures. This assumption seemed reasonable because due to the relatively small amount

of added coal (i.e., relatively smaller surface area) to the soil mixtures relative to the amount of the soil medium (i.e., relative larger surface area), the effect of the coal on the properties of the soil medium would be negligible. For example, in all soil mixtures with 0.5wt% coal content, for each gram of coal in the mixture there were approximately 199 grams of soil material. Similarly, there were approximately 99 g of soil for 1 g of coal in the 1.0%-coal samples, and approximately 39 g for 1 g of coal in the 2.5%-coal samples, approximately 19 g for 1 g of coal in the 5% coal samples, and approximately 9 g for 1 g of coal in the 10% coal samples. Based on the relatively small amount of coal addition to the soil mixtures compared to the soil medium, the underlying assumption for computing the A ratio, per soil material, is that the effect of coal on the rate of carbon loss from SOM in the soils would be negligible.

Depicted in Figure 1-1-4 are the data used to compute ratio A for mine soils derived from SS, SiS, 50:50 SS:SiS mixture, and the reference quartz sand and Jefferson forest soil materials. The following list depicts the estimates for ratio A used in Eq.1 and Eq.2 for the above five materials:

Material	Ratio A
Sandstone (1)	4.6835
Siltstone (2)	12.7343
50:50 SS:SiS (3)	11.6941
Quartz sand (4)	7.5277
Jefferson soil (5)	16.2548

Anchored in the above-mentioned examples for the amount of coal relative to the amount of soil material present in the soil mixtures, and based on the assumption for computing ratio A, we expected that the soil medium would have a considerable effect on the rate of carbon loss from the coal present in the soil mixtures. This is to say that the rate of carbon loss from pure coal material could be substantially different than that from the same amount of coal “surrounded” by soil particles in a given soil medium.

In order to analyze the effect of soil material on the thermal properties of coal in the soil mixtures, we performed an analysis of variance (ANOVA) of our data where the dependent variable was G_{375} , and material type and coal content were the two independent variables. The G_{375} variable was computed as the difference $[C_{375 \text{ for } Z\% \text{ coal}} - C_{375 \text{ for } 0\% \text{ coal}}]$, where $Z\%$ represents the coal content level in each sample from the data depicted in Figure 1-1-3.

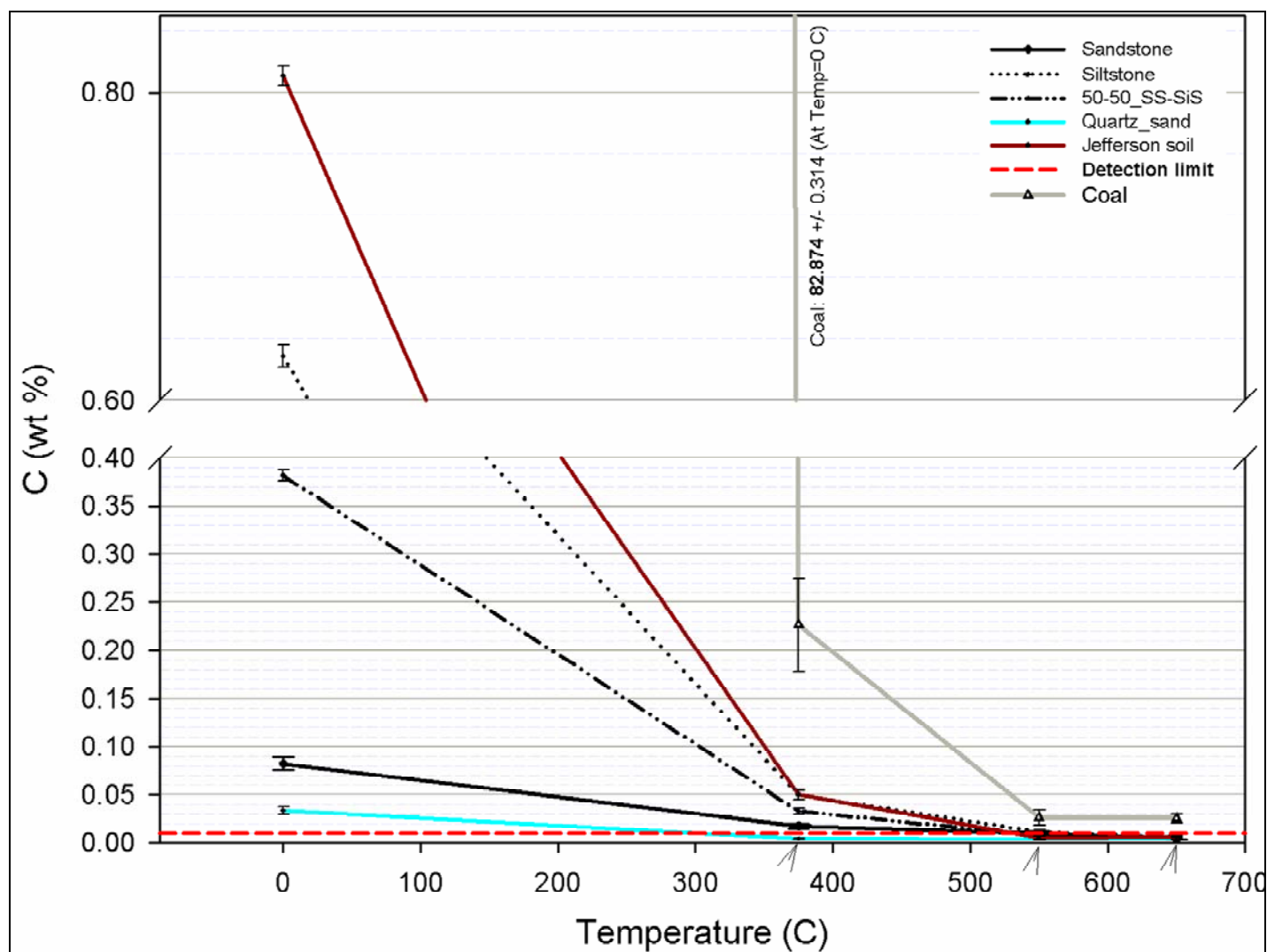


Figure 1-1-4. Average carbon concentration estimates for six coal-free mine soil materials, SS, SiS, 50:50-mixture of SS and SiS, quartz sand, Jefferson forest soil, and soil-free coal for three thermal treatment temperatures, 375, 550, and 650°C. Vertical lines represent the standard error of the mean. The horizontal dash-line depicts the detection limit (0.01 wt %) of the carbon auto-analyzer machine used in this study.

The result from the ANOVA analysis (N = 123 observations for 5 levels of material type and 5 levels of coal content: 0.5, 1, 2.5, 5, 10) showed that material type ($P = 0.0133$), coal content ($P < 0.0001$), and the interaction between material type and coal content ($P = 0.0003$) were significantly affecting the magnitude of G_{375} . Results from a similar ANOVA analysis for G_0 as the dependent variable, and material type and coal content as the two independent variables, showed that material type ($P = 0.0065$) and coal content ($P < 0.0001$) were significantly affecting the magnitude of G_0 . The effect of the interaction between material type and coal content ($P = 0.2857$) on the magnitude of G_0 was not significant. The dependent variable G_0 was computed as the difference [C_0 for $Z\%$ coal – C_0 for 0% coal], where $Z\%$ represents the coal content level in each sample from the data in Figure 1-1-3. The above-mentioned ANOVA analyses for G_0 and G_{375} indicated that material-specific ratios $B (= G_0/G_{375})$ were necessary for each of the five soil materials tested.

Because the relative content of coal in the soil samples significantly affected the thermal properties of the coal, we performed regression analyses (PROC REG in SAS® statistical software) to model the G_{375} for each sample as a function of the G_{375} of pure coal, which was 0.226 wt% (Figure 1-1-3). Prior to the modeling task, the G_{375} carbon content measured after a 375°C thermal pre-treatment was adjusted to the coal content level of each sample. For example, G_{375} of the pure coal was multiplied by a factor of 0.5/100 to match all soil samples with 0.5wt% coal content. Similarly, the G_0 of each sample was modeled as a function of the G_0 of pure coal, (82.874 wt% C) (Figure I-4). The G_0 carbon content was also adjusted to the coal content level of the samples prior to modeling.

In order to stabilize the variance of the residuals and meet the normality assumption of the dependent variable in these regression analyses, the G_0 and G_{375} variables were transformed using the square root transformation, i.e., $G_0' = \text{SQRT}(G_0)$ and $G_{375}' = \text{SQRT}(G_{375})$. The Cp-model selection procedure in SAS® was used to determine the most appropriate model that would characterize G_0' and G_{375}' as a function of the adjusted G_0 and G_{375} , respectively, of pure coal. For all models, the intercept was forced through 0 because extremely small levels of coal content in the soil would result in a very small surface area relative to total surface area of the soil sample; hence, the effect of the soil medium on the coal will be infinitely small. Table 1-1-1 depicts the regression models that were developed to estimate G_0 and G_{375} of the soil samples of each material type as a function of the adjusted G_0 and G_{375} of pure coal.

Table 1-1-1. Regression models used to estimate G_0 and G_{375} of soil samples as a function of the G_0 and G_{375} of pure coal. Ratio B was used to calculate pedogenic C (wt%) and geogenic C (wt%) in Eq.1 and Eq.2, respectively.

Material Type	Dependent Variable	a	n	R^2	R^2	Adj.-	Ratio B
(1) Sandstone	$\sqrt{G_0}$	0.99535	25	99.99	99.99	* F [@]	0.70359
	$\sqrt{G_{375}}$	1.18663	20	88.14	87.52		
(2) Siltstone	$\sqrt{G_0}$	1.00605	24	99.99	99.99	* F	0.22158
	$\sqrt{G_{375}}$	2.13724	24	94.36	94.11		
50:50 SS:SiS (3)	$\sqrt{G_0}$	0.99613	25	99.98	99.98	* F	0.41394
	$\sqrt{G_{375}}$	1.54827	25	91.13	90.76		
Quartz sand (4)	$\sqrt{G_0}$	0.99952	24	99.97	99.97	* F	0.37108
	$\sqrt{G_{375}}$	1.64081	23	91.48	91.10		
Jefferson soil (5)	$\sqrt{G_0}$	0.99519	25	99.99	99.99	* F	0.53876
	$\sqrt{G_{375}}$	1.35584	18	52.76	49.98		

For each soil material, there were five replicated measurements of C_0 and C_{375} for five levels of coal content, 0.5, 1, 2.5, 5, and 10 wt%. The regression models used to estimate the G_0 were developed from a total of 25 measurements per material type. All 25 G_0 estimates were used in the models with the exception of the quartz sand and SiS materials. One outlier each was removed from the SiS and quartz data sets for this modeling task (Table 1-1-1). The regression models used to estimate the G_{375} were developed using all non-negative G_{375} estimates per material type. The negative G_{375} estimates implied that in some sample replications, the majority of which had the lowest level of coal content, the added coal was completely oxidized to CO_2 following the 375°C pretreatment. As mentioned above, G_{375} was estimated as the difference [$C_{375 \text{ for } Z\% \text{ coal}} - C_{375 \text{ for } 0\% \text{ coal}}$]. For example, the carbon content following 375°C pretreatment (i.e. $C_{375 \text{ for } Z\% \text{ coal}}$) of five SS sample replications (three with 0.5wt% coal and two with 1.0wt% coal) was less than the average carbon content of five SS samples replications with no coal (i.e., $C_{375 \text{ for } 0\% \text{ coal}}$) following 375°C pretreatment (Table 1-1-1).

The estimates for ratio A, presented above, could be used for other mined sites within the Appalachian coalfield to determine pedogenic carbon and GC provided that the mine spoil material is of SS, SiS, or 50:50 SS:SiS origin. Because coal quality may vary across mined sites, resulting in significantly different thermal properties of coal fragments in the soil medium, the estimates for ratio B used in Eq. 1 and Eq. 2 could be computed from the coefficients presented in Table 1-1-1 and the coal quality-specific property F ($=G_0/G_{375}$). For example, for SS mine spoil material, B is computed as $0.70359 \cdot F$.

Accuracy and Precision of the 16-Step SOC Estimation and the Walkley-Black Methods for Mine Soils

Estimates for pedogenic carbon content for each of the 30 soil mixtures, also referred to as SOC%_{estimated} in this text, were computed using Eq. 1. The Walkley-Black organic carbon estimates were computed as described in the Methods section of this section.

In order to evaluate the accuracy of the 16-step SOC estimation and Walkley-Black methods, PRUM (%) was estimated for the individual sample replications made by the 16-step method (5 replications) and for the Walkley-Black procedure (3 replications). Additionally, for the 30 soil mixtures, PRUM_{5reps} (%) was evaluated for the SOC%_{estimated} by the 16-step method where C₀ and C₃₇₅ (used in Eq. 1) were computed as the average of five replicated C₀ and C₃₇₅ measurements. Averaging of the replicated C₀ and C₃₇₅ measurements was done to determine whether a larger number of replicated measurements for C₀ and C₃₇₅ would increase the accuracy of SOC%_{estimated}. Although in most cases increasing the number of measurements usually leads to a better final estimate of a soil property, soil scientists have to make a decision whether the relative gain in accuracy of the estimate is worth the additional costs associated with carrying out additional measurements.

The precision of a certain measurement method describes the repeatability of the end results. In order to determine the precision of the two methods, the standard error of PRUM (%), which is a measure of the relative accuracy of the results, was estimated (Tables 1-1-2 and 1-1-3). Using the mean and standard error of PRUM (%), the confidence limits of the accuracy of SOC%_{estimated} by the 16-step SOC estimation method (Table 1-1-2) and by the Walkley-Black procedure (Table 1-1-3) could be estimated for three probability levels, 80%, 90%, and 95%, using the corresponding student's *t* values from a statistical *t*-table for 4 degrees of freedom for the 16-step method, and for 2 degrees of freedom for the Walkley-Black procedure. There were no standard error estimates computed for the mean PRUM_{5reps}(%) in Table 1-1-2. These values were derived from individual C₀ and C₃₇₅ measurements (average of five C₀ and C₃₇₅ replicated measurements), as opposed to the mean PRUM (%), which was estimated as the average PRUM (%) of five individual measurement (Table 1-1-2).

Due to the design of the Walkley-Black procedure, each individual measurement of a total of three measurement replications was analyzed separately from the rest and was associated with a unique PRUM (%) estimate. For all parameters, other than the blank, there was no valid way to combine the replicated measurements done in a Walkley-Black procedure (such as the volume of the sample titer, which is specific for the amount of oven-dried soil used for this one replication) to calculate a better average estimate that could then be used in the equation to produce a better Walkley-Black organic carbon estimate (Table 1-1-3).

The results for PRUM (%) of estimated SOC values from both methods (Tables 1-1-2 and 1-1-3, Fig. 1-1-5) indicated that neither method was suitable to predict SOC of quartz sand mixtures used in this study. The PRUM (%) for the quartz sand mixtures with added coal ranged from 38% to 412% for the 16-step method, and from 98% to 2,704% for the Walkley-Black method, indicating a colossal over-estimation of the SOC present in these mixtures, which was 0.033 C wt%. The main reason for this is attributed to the problem associated with handling quartz sand mixtures, and specifically, the electrostatic interactions between the soil particles and the plastic storage bag. Because all samples were stored in plastic bags between measurements, there was a clearly observable electrostatic interaction between the coal particles and the bag, as

well as between the individual quartz particles and the inside surface of the plastic bag. As a result, each time the soil mixture was measured, unspecified amounts of coal and sand were electrostatically removed from the sample and were adsorbed to the inside surface of the plastic bag. For these reasons, the results for quartz sand mixtures will not be further discussed.

The mean PRUM (%) of SOC estimates made by a single replication of the 16-step method ranged from 13% to 148% for SS (containing SOC at 0.082 C wt%), from 11% to 41% for SiS (SOC at 0.629 C wt%), from 5% to 84% for the 50:50 SS:SiS mixture (SOC at 0.382 C wt%), and from 9% to 103% for the Jefferson forest soil (SOC at 0.811 C wt%), across the spectrum of coal content levels (Table 1-1-2, Fig. 1-1-5).

Table 1-1-2. Percent relative uncertainty from the mean (PRUM, %) of the 16-step SOC estimation method for 30 mine soil mixtures constructed from 5 soil materials at 6 levels of coal content. Five replicated SOC measurements were made for each mixture. The right-most column depicts the PRUM_{5reps} (%), described in the text.

Material	Coal Content	Mean	Std.Error	MIN	MAX	n	MEAN 5reps
	wt %	----- % -----					%
Sandstone (1)	0	16.9	1.4	6.1	46.4	5	0.0
	0.5	14.6	1.1	12.0	19.2	5	2.3
	1	12.6	1.5	3.7	28.4	5	8.3
	2.5	13.0	1.6	2.3	29.6	5	8.9
	5	23.8	1.1	19.0	30.6	5	14.8
	10	148.0	1.3	76.6	337.2	5	60.1
Siltstone (2)	0	21.3	1.2	13.2	32.1	5	0.0
	0.5	10.9	2.1	0.6	25.2	5	18.3
	1	15.5	1.9	1.2	47.7	5	14.5
	2.5	33.2	1.6	5.5	90.7	5	31.5
	5	40.9	1.3	16.8	92.9	5	25.0
	10	15.8	1.5	5.1	44.6	5	16.4
50-50_SS-SiS (3)	0	10.4	1.7	1.7	36.3	5	0.0
	0.5	6.9	1.5	1.8	17.9	5	7.5
	1	8.6	2.1	0.7	46.0	5	19.0
	2.5	5.4	1.3	2.6	10.9	5	2.0
	5	34.4	1.2	19.7	54.8	5	36.5
	10	84.4	1.5	29.6	277.9	5	45.3
Quartz_sand (4)	0	21.2	1.3	8.9	32.6	5	0.0
	0.5	38.1	1.2	22.7	61.9	5	40.9
	1	84.8	1.3	39.6	152.7	5	35.9
	2.5	309.9	1.2	204.8	511.9	5	59.7
	5	302.2	1.3	114.7	485.2	5	38.8
	10	412.1	1.5	182.2	1839.9	5	199.1
Jefferson_soil (5)	0	20.1	1.2	13.5	30.8	5	0.0
	0.5	11.0	1.7	1.4	25.2	5	12.0
	1	8.8	1.2	4.1	14.7	5	7.8
	2.5	27.9	1.8	9.0	229.4	5	30.9
	5	103.4	1.7	19.4	535.8	5	172.6
	10	65.7	1.2	39.2	86.7	5	68.1

Table 1-1-3. Percent relative uncertainty from the mean (PRUM, %) of the Walkley-Black organic carbon measurements for 30 mine soil mixtures constructed from five soil materials at six levels of coal content. Three replicated measurements were made for each mixture.

Material	Coal Content	Mean	Std.Error	MIN	MAX	n
	wt %	----- % -----				
Sandstone (1)	0	51.1	1.3	32.6	66.5	3
	0.5	17.3	1.3	12.1	29.3	3
	1	98.5	1.2	72.6	126.3	3
	2.5	203.3	1.3	125.9	343.1	3
	5	427.3	1.2	288.3	547.2	3
	10	1002.8	1.2	694.5	1412.3	3
Siltstone (2)	0	16.7	1.1	15.3	19.6	3
	0.5	22.3	2.3	8.7	117.6	3
	1	2.1	1.4	1.2	4.1	3
	2.5	30.6	1.4	17.0	45.8	3
	5	51.9	1.2	38.5	65.4	3
	10	122.1	1.1	105.9	139.4	3
50-50_SS-SiS (3)	0	15.6	1.7	6.2	41.0	3
	0.5	12.9	1.1	10.9	14.8	3
	1	3.3	1.4	2.0	5.8	3
	2.5	40.4	1.1	33.4	50.0	3
	5	63.6	1.4	35.8	102.0	3
	10	125.8	1.5	55.2	204.1	3
Quartz_sand (4)	0	5.3		5.3	5.3	3
	0.5	98.0	1.5	45.7	147.6	3
	1	279.1	1.1	227.8	356.8	3
	2.5	512.8	1.2	338.1	715.0	3
	5	1247.8	1.2	1019.2	1674.2	3
	10	2703.6	1.1	2402.1	2948.6	3
Jefferson_soil (5)	0	2.2	1.4	1.4	4.2	3
	0.5	2.7	1.6	1.3	7.0	3
	1	6.9	1.1	5.4	8.0	3
	2.5	20.0	1.4	13.0	41.7	3
	5	7.9	3.1	0.8	27.0	3
	10	57.8	1.1	47.4	66.1	3

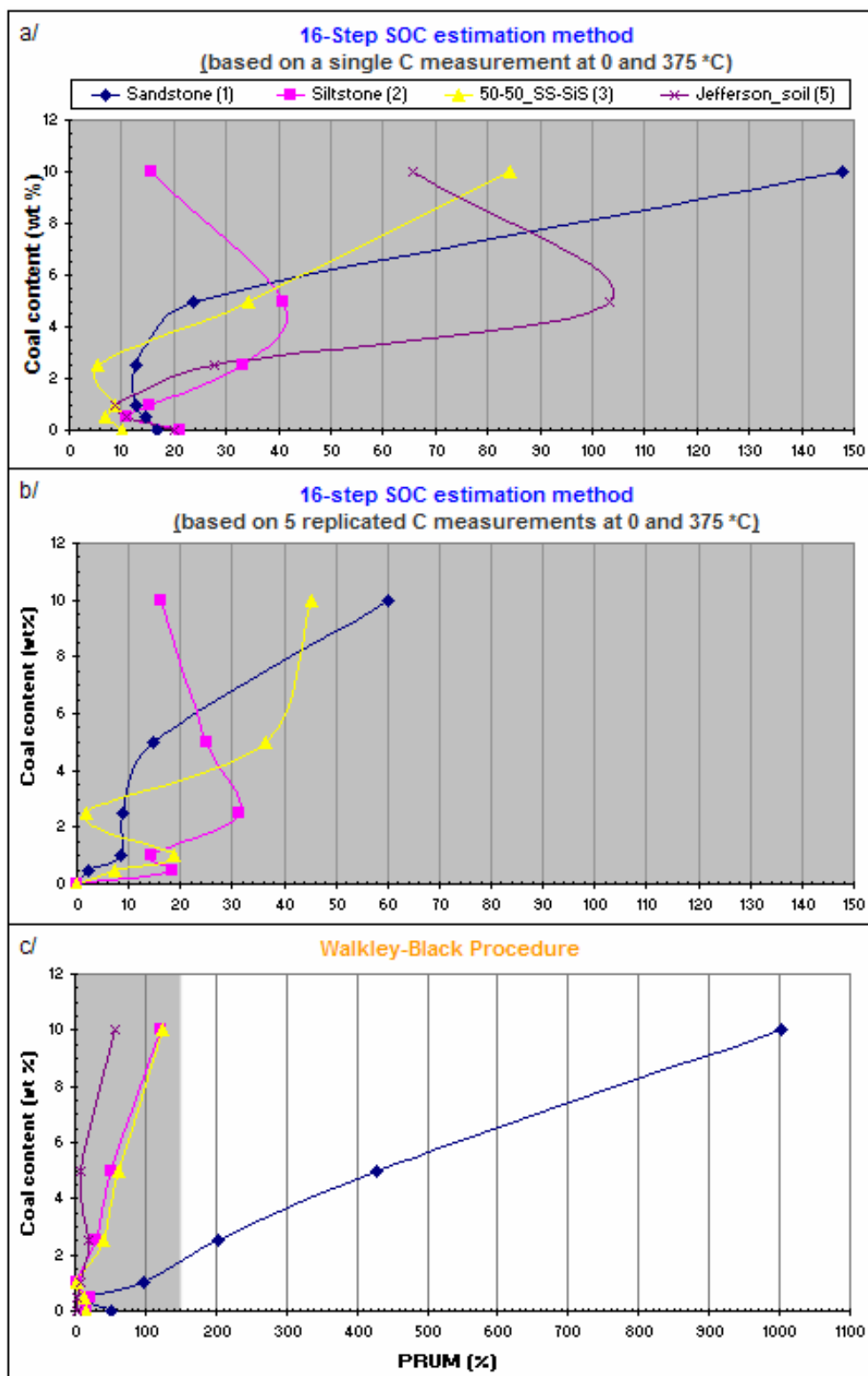


Figure 1-1-5. Average percent relative uncertainty from the mean (PRUM, %) for SOC measurements made by the 16-step estimation method (a/ and b/), and by the Walkley-Black procedure (c/). The gray area in graph c/ is equivalent to the full extent of graphs a/ and b/.

The corresponding PRUM (%) estimates of SOC predictions made by the Walkley-Black procedure were equal to or greater than those by the 16-step procedure for all materials except for the Jefferson forest soil. The mean PRUM (%) for the Walkley-Black estimates ranged from 17% to 1,003% for SS, from 2% to 122% for SiS, from 3% to 126% for the 50:50 SS:SiS, and from 2% to 58% for the Jefferson forest soil (Table 1-1-3, Fig. 1-1-5).

The Walkley-Black procedure yielded better SOC estimates for the Jefferson forest soil mixtures compared to the 16-step method. The latter could be due to the fact that the Walkley-Black method was specifically calibrated for natural, undisturbed soils such as the Jefferson forest soil (Tables 1-1-2 and 1-1-3, Fig. 1-1-5). Furthermore, the increase of coal content in the Jefferson forest soil mixture did not result in a dramatic increase in the uncertainty of the SOC estimates made by the Walkley-Black method relative to all other materials (Table 1-1-3). However, for the rest of the materials, an increase of the coal content in the soils resulted in an appreciable increase in the uncertainty associated with the SOC predictions made by the Walkley-Black procedure.

The overall results indicated that the 16-step SOC estimation method yielded better results (with lower PRUM %) than the Walkley-Black procedure for all mixtures constructed from SS, SiS, and the 50:50 SS:SiS materials (Tables 1-1-2 and 1-1-3) (Fig. 1-1-5). The SS, SiS, and the combination of SS-SiS materials are also the most likely soil medium for mine soil development in the hard-rock regions of the Appalachian coalfield. Therefore, we expect that the 16-step SOC estimation method would be the preferred method for SOC estimation on mined land in this region.

The accuracy of SOC estimates for the SS material by the 16-step method increased by increasing the number of measurement replications, but did not appreciably improve for the SiS and the 50:50 SS:SiS mixtures (Table 1-1-2, Fig. 1-1-5). This suggested that measuring SOC in mine soils of SS and SiS origin could be possible from a single measurement of C_0 and C_{375} (Eq.1), and that additional replications may not improve the accuracy of these measurements.

In order to determine the effects of material type and coal content in the soil on the accuracy of SOC estimation in mine soils, we performed ANOVA analyses using the PRUM (%) data (dependent variable) of the SOC estimates from the 16-step and the Walkley-Black methods. Only data for SS, SiS, and the 50:50 SS:SiS mixtures were used in the ANOVA for the 16-step method as these materials were the targeted soil media for evaluation that could be used for carbon projects on mined lands. Data for all soil materials, excluding quartz sand, were used in the ANOVA analysis for the Walkley-Black procedure.

The results for the 16-step method indicated that the effects of coal content ($P < 0.0001$) and the interaction between material type and coal content ($P = 0.0146$) were significant, but material type ($P = 0.1798$) was not significant. The results for the Walkley-Black procedure showed that the effects of material type ($P < 0.0001$), coal content ($P < 0.0001$), and their interaction ($P = 0.0008$) were significant. Because of the significant effect of the interaction between the independent variables for both methods, means-separation analyses (PROC GLM in SAS®; multiple comparison of least squares means; adjustment used - Tukey) were used to determine which means, by specific material type and by coal content level, were significantly different from the rest (Tables 1-1-4 and 1-1-5).

Table 1-1-4. Multiple comparison of the means of the log-transformed PRUM(%) of SOC estimates from the 16-step method for three material types, SS, SiS, and 50:50 SS:SiS, and six levels of coal content, 0, 0.5, 1, 2.5, 5, and 10wt%. Highlighted rows and columns in the list (left) and the table (right) depict the means that were significantly different from the rest.

Material	Coal Content	ID	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18
50:50 SS:SiS	0	1				*						***								
	1	2	1.00			**						***								
	5	3	0.88	0.70																
	10	4	0.09	0.04	0.99		**	***												
	0.5	5	1.00	1.00	0.45	0.01						***								
	2.5	6	1.00	1.00	0.22	<0.01	1.00					***								
Sandstone	0	7	1.00	1.00	1.00	0.45	0.99	0.92				*								
	1	8	1.00	1.00	0.97	0.18	1.00	1.00	1.00			**								
	5	9	1.00	0.97	1.00	0.83	0.85	0.60	1.00	1.00										
	10	10	<0.01	<0.01	0.62	1.00	<0.01	<0.01	0.06	0.02	0.24		**	**		**		**	***	
	0.5	11	1.00	1.00	0.99	0.30	1.00	0.98	1.00	1.00	1.00	0.03								
	2.5	12	1.00	1.00	0.98	0.20	1.00	0.99	1.00	1.00	1.00	0.02	1.00							
Siltstone	0	13	1.00	0.99	1.00	0.72	0.92	0.73	1.00	1.00	1.00	0.16	1.00	1.00						
	1	14	1.00	1.00	1.00	0.36	1.00	0.96	1.00	1.00	1.00	0.04	1.00	1.00	1.00					
	5	15	0.72	0.51	1.00	1.00	0.27	0.12	0.99	0.90	1.00	0.81	0.96	0.91	1.00	0.98				
	10	16	1.00	1.00	1.00	0.38	1.00	0.95	1.00	1.00	1.00	0.05	1.00	1.00	1.00	1.00	0.98			
	0.5	17	1.00	1.00	0.91	0.10	1.00	1.00	1.00	1.00	1.00	<0.01	1.00	1.00	1.00	1.00	0.77	1.00		
	2.5	18	0.90	0.74	1.00	0.99	0.49	0.25	1.00	1.00	0.98	1.00	1.00	0.98	1.00	1.00	1.00	1.00	0.93	

ID = sample identification number;

* significantly different at the 0.10 level;

** significantly different

at the 0.05 level;

*** significantly different at the 0.01 level;

Table 1-1-5. Multiple comparison of the means of the log-transformed PRUM(%) of SOC estimates from the Walkley-Black procedure for three material types, SS, SiS, and 50:50 SS:SiS, and six levels of coal content, 0, 0.5, 1, 2.5, 5, and 10wt%.

Material	Coal Content	ID	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24
50:50 SS:SiS	0	1				*											***	***		***				*		
	1	2	0.48		***	***		***				***			***	***	***	***		***			***	***		**
	5	3	0.67	<0.01					***	**	*		***					***				***				
	10	4	0.07	<0.01	1.00		**		***	***	***		***					*			*	***				
	0.5	5	1.00	0.71	0.43	0.03										*	***	***		***				**		
	2.5	6	0.99	<0.01	1.00	0.92	0.92		***				***				**	***				***				
Jefferson soil	0	7	0.12	1.00	<0.01	<0.01	0.26	<0.01				***		**	***	***	***	***	*	***	*		***	***	**	***
	1	8	1.00	1.00	0.04	<0.01	1.00	0.26	0.92			*				***	***	***		***		*	***			
	5	9	1.00	0.99	0.07	<0.01	1.00	0.40	0.81	1.00						***	***	***		***			***			
	10	10	0.78	<0.01	1.00	0.55	1.00	0.40	<0.01	0.06	0.11		***					***				***				
	0.5	11	0.28	1.00	<0.01	1.00	0.49	<0.01	1.00	0.99	0.96	<0.01			***	***	***	***		***			***	***	*	**
	2.5	12	1.00	0.22	0.91	0.20	1.00	1.00	0.04	0.96	0.99	0.96	0.11				***	***		**		**				
Sandstone	0	13	0.89	<0.01	1.00	0.99	0.70	1.00	<0.01	0.11	0.18	1.00	<0.01	0.99			*	***				***				
	1	14	0.20	<0.01	1.00	1.00	0.09	0.99	<0.01	<0.01	<0.01	1.00	<0.01	0.44	1.00			**				***				
	5	15	<0.01	<0.01	0.15	0.86	<0.01	0.02	<0.01	<0.01	<0.01	0.10	<0.01	<0.01	0.06	0.59			***		***	***	*		***	***
	10	16	<0.01	<0.01	<0.01	0.08	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0.02	0.02	1.00		***		***	***	***	*	***	***
	0.5	17	1.00	0.36	0.79	0.11	1.00	1.00	0.08	0.99	1.00	0.88	0.19	1.00	0.95	0.28	<0.01	<0.01		**		*				
	2.5	18	<0.01	<0.01	0.91	1.00	<0.01	0.41	<0.01	<0.01	<0.01	0.83	<0.01	0.02	0.70	1.00	1.00	0.43	0.01		***	***		**		
Siltstone	0	19	1.00	1.00	0.75	0.10	1.00	<0.01	0.09	1.00	1.00	0.85	0.22	1.00	0.93	0.25	<0.01	<0.01	<0.01			*				
	1	20	0.11	1.00	<0.01	<0.01	0.22	<0.01	1.00	0.89	0.76	<0.01	1.00	0.03	<0.01	<0.01	<0.01	<0.01	0.07	<0.01	0.08		***	***	**	***
	5	21	0.88	<0.01	1.00	0.99	0.69	1.00	<0.01	0.10	0.17	1.00	<0.01	0.99	1.00	1.00	0.07	<0.01	0.94	0.72	0.92	<0.01				
	10	22	0.08	<0.01	1.00	1.00	0.03	0.94	<0.01	<0.01	<0.01	1.00	<0.01	0.22	1.00	1.00	0.84	0.07	0.13	1.00	0.11	<0.01	1.00			
	0.5	23	1.00	0.15	0.96	0.29	1.00	1.00	0.02	0.91	0.97	0.99	0.07	1.00	1.00	0.57	<0.01	<0.01	1.00	0.04	1.00	0.02	1.00	0.32		
	2.5	24	1.00	0.04	1.00	0.66	1.00	1.00	<0.01	0.57	0.74	1.00	0.01	1.00	1.00	0.90	<0.01	<0.01	1.00	0.16	1.00	<0.01	1.00	0.70	1.00	

ID = sample identification number;

* significantly different at the 0.10 level; ** significantly different at the 0.05 level; *** significantly different at the 0.01 level;

The dependent variable for the means-separation analyses was the log-transformed PRUM(%), and the independent variables were material type and coal content. Log-transformation ($\text{PRUM}'(\%) = \ln(\text{PRUM}(\%))$) was required to satisfy the normality and the constant variance assumptions needed for an ANOVA analysis. Although the arcsin transformation ($y' = \sin^{-1}(\sqrt{y})$) (Montgomery et al., 2001, p.174) is usually recommended for proportion-type variables, this transformation failed to stabilize the variance of the residuals for our data.

The means-separation analysis for the 16-step method indicated that the 10wt%-coal samples for the SS and 50:50 SS:SiS materials were the only samples with significantly higher PRUM(%) compared to the remaining 28 mixtures (Table 1-1-4). These results provided sufficient evidence to claim that the 16-step SOC estimation method produced consistent SOC estimates (with consistent relative uncertainty) for the three most common mine spoil materials tested, with coal contents ranging from 0 to 10 wt%, with the exception of the 10%-coal levels of the SS and the 50:50 SS:SiS samples. The PRUM(%) of five replications of the SS mixture with 10% coal level ranged from 77 to 337, and the PRUM(%) of five replications of the 50:50 SS:SiS mixture with 10% coal level ranged from 30 to 278 (Table 1-1-2).

In order to achieve better SOC accuracy and precision (i.e., lower PRUM, %) for soil samples of SS and 50:50 SS:SiS material type at high levels of coal content, replicated measurements of C_0 and C_{375} could be made prior to SOC computation. The data indicated that for the SS and 50:50 SS:SiS material types at 10wt% coal content, using five replications would produce SOC estimates with PRUM at 60 and 45%, respectively (Table 1-1-2).

However, the same could not be claimed for the Walkley-Black procedure. The means-separation analysis results indicated that the performance of the Walkley-Black procedure for SOC estimation varied across material type and across coal content level without any apparent trends (Table 1-1-5). The latter supported the claims by other researchers showing that SOC measurements made by the Walkley-Black procedure could be of appreciable uncertainty depending on the properties of the soil (Daniels and Amos, 1982; Skjemstad and Taylor, 1999).

Effects of Soil Material Type and Coal Content on Walkley-Black Soil Carbon Measurements

The third objective of this project was to determine the effects of material type and coal content on the accuracy of Walkley-Black soil carbon estimates for mine soils. Furthermore, if there were a significant correlation between the SOC content present in the mine soil mixtures and their respective Walkley-Black carbon estimates, material type and coal content, then we would be able to develop a model that would allow Walkley-Black SOC values for mine soils estimated by the procedure in the past to be corrected for induced uncertainties due to coal content that are usually associated with Walkley-Black estimates.

Based on the results from the ANOVA analysis for PRUM(%) described in the previous section, it was evident that material type and coal content significantly affected the uncertainties associated with SOC estimates made by the Walkley-Black method (Table 1-1-5); hence, the SOC predictions were affected by material type and coal content. Because any potential regression models to correct Walkley-Black carbon estimates would be used for mine soils only, we developed the correction procedure based on the data for the SS, SiS, and 50:50 SS:SiS materials. The SOC content present in these three materials ($\text{SOC}\%_{\text{present}}$), as the dependent variable, was regressed (PROC REG in SAS®) against the SOC estimates made by the Walkley-Black method and material type, as the two independent variables (Fig. 1-1-6).

Walkley-Black SOC measurement correction for coal:

$$\text{SOC (wt\%)} = 0.11063 * (\text{Order}) + 0.28687 * (\text{WB-SOC, \%}); \quad R^2=54.72, n=27, P>0.0001.$$

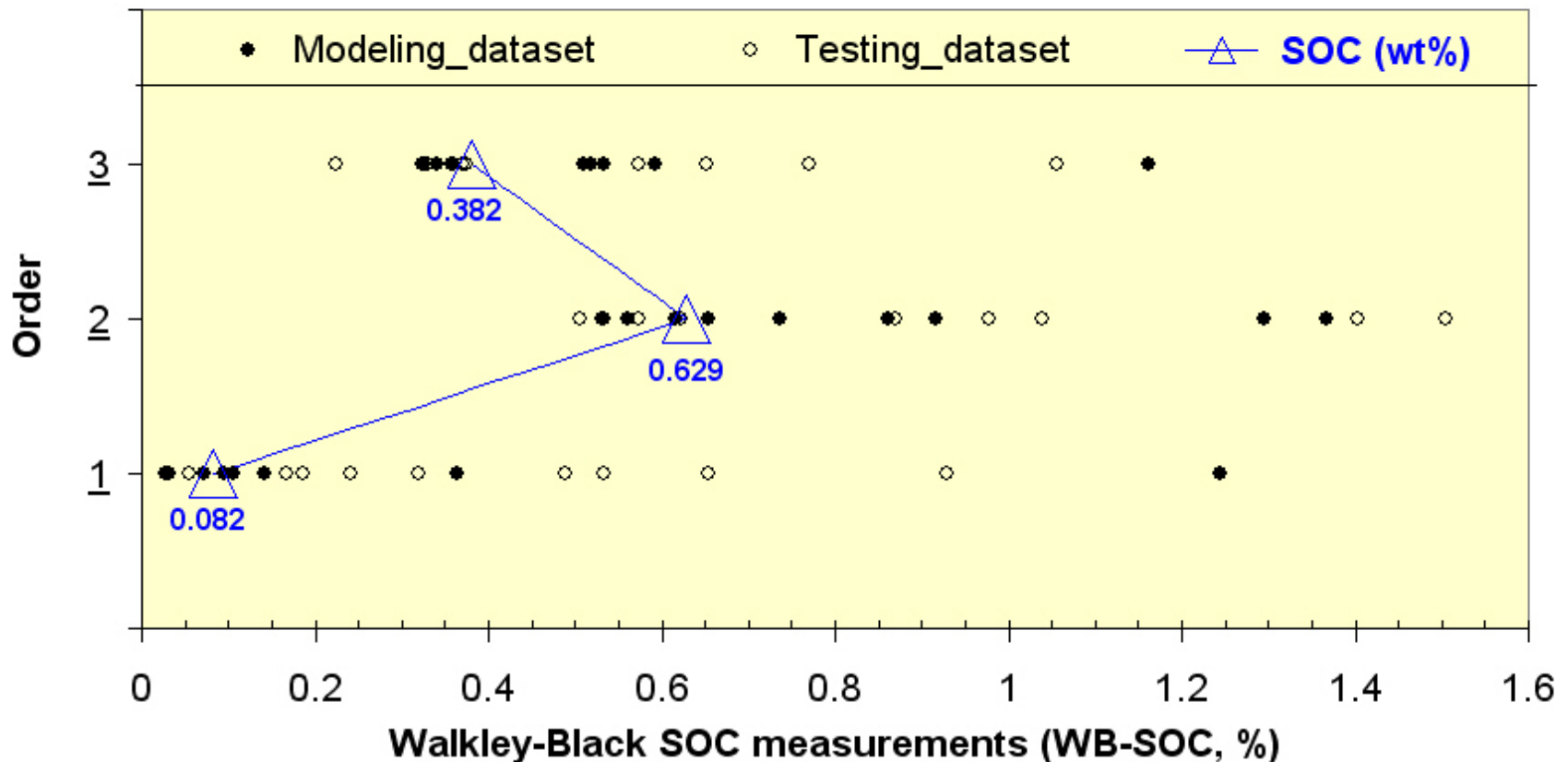


Figure 1-1-6. Development and testing of a regression model for coal-correction of Walkley-Black carbon measurements in mine soils derived from SS (Order=1), SiS (Order=2) and 50:50 SS:SiS (Order=3) spoil materials. The SOC content, as percent by sample weight, present in each material is represented by the triangles, SOC (wt%).

The R^2 of the model indicated that 55% of the uncertainty associated with measuring the $SOC\%_{\text{present}}$ by the Walkley-Black procedure was explained by material type; i.e., the variable *order*, and by the WB-SOC% data for the 54 samples analyzed (a total of three replications of three materials at six coal content levels) (Figure 1-1-6). The final regression model for coal-correction of Walkley-Black carbon estimates for mine soils was developed as follows:

$$SOC(wt\%) = 0.11063 * (Order) + 0.28687 * (WB - SOC, \%) \quad \text{Eq.3}$$

where SOC (wt%) = sample SOC content expressed as percent by sample weight; Order = 1 for SS, 2 for SiS, and 3 for 50:50 SS:SiS material; WB-SOC, % = Walkley-Black SOC estimate.

Although a certain fraction of the variation of Walkley-Black carbon estimates for mine soils remained unexplained, the results of this study indicated that the majority of the variation is due to the oxidation of carbon from coal fragments. This was evident from the fact that our model explained 55% of the variation of Walkley-Black soil carbon estimates and the remainder 45% was unexplained. We believe that the unexplained fraction of the variation could be due to the cumulative effect of ferrous iron, chlorides, and higher oxides of manganese, all of which could be present in young mine soils and could interfere with the Walkley-Black procedure (Nelson and Sommers 1982).

By substituting 1, 2, and 3 for variable *order* in Eq. 3, for SS-, SiS-, and 50:50 SS:SiS-derived mine spoil materials, respectively, mined land researchers would be able to use their Walkley-Black soil carbon estimates from the past to produce a better approximation of the true SOC present in the samples for their respective study areas. For mine soils derived from spoil material of ratios between SS and SiS that are different than 50:50, separate analyses similar to the one presented in this study should be performed to develop valid models for Walkley-Black soil carbon estimate correction.

It should be noted that the coal-correction equation (Eq. 3) was based on the modeling and model-testing analyses of standard mine soil samples prepared with bituminous coal fragments that have approximate carbon content of 83wt%. Although Eq. 3 would not be appropriate for coal-correction of Walkley-Black estimates for mined lands where, instead of bituminous coal, fragments of different coal type (or rank) dominate the soils, the methods used in this paper could be used to develop a new, appropriate coal-correction formula.

The distributions of the Walkley-Black estimates for the SS, SiS, and the 50:50 SS:SiS materials are shown in Figure 1-1-6. The results indicated that the majority of the WB-SOC(%) values for the three materials tend to overestimate the SOC content in the soil samples and only a small number of these values underestimated the SOC content. The deviation of the Walkley-Black estimates from the SOC content present in the samples increased as the relative coal content increased (Table 1-1-3). Because higher oxides of manganese in the soil could compete for SOC with the oxidizing agent used in the Walkley-Black procedure (under acidic conditions) thus causing the resulting SOC values to underestimate the SOC content in the soil, the results in Figure 1-1-6 suggested that some of the soil materials tested could have appreciable amounts of manganese oxides.

There are certain benefits in developing site-specific, coal-correction equations for Walkley-Black soil carbon values obtained in the past. On one hand, before now, the Walkley-Black procedure was commonly used for SOC estimation in mine soils (Akala and Lal 2001b;

Amichev et al. 2004; Rodrigue 2001) assuming no interferences from coal fragments or from the spoil material. Additionally, quite often these soil samples were archived for a certain period of time after which they were discarded. Therefore, in order to maintain the collected carbon data, and to determine and monitor carbon sequestration on mined lands in the long term, coal-correction equations for Walkley-Black soil carbon estimates similar to Eq.3 could prove useful and cost-effective. On the other hand, using coal-correction for Walkley-Black carbon estimates from past studies would lead to carbon estimates that are much closer to the true SOC content in these mine soils. The direct implication of having better SOC estimates on mined land is that, now, it would be possible to compare the carbon sequestration potential of mined lands with other terrestrial ecosystems of different land use and management regimes.

There are some weaknesses associated with developing coal correction equations for Walkley-Black soil carbon values that are based only on spoil type and the carbon estimates obtained in the past. Assuming that the coal content of a mine soil remained constant for a long period of time, and it was measured with certain accuracy and precision by the 16-step method, other physicochemical properties of mine soils would change with time. As mine soils age, the spoil material is quickly weathered, both mechanically and chemically (Haering et al. 1993). As a result of overburden weathering, along with changes in other physicochemical properties, the Fe^{2+} (ferrous iron) is oxidized to Fe^{3+} (ferric iron), meaning that in older mine soils the $\text{Fe}^{2+}/\text{Fe}^{3+}$ ratio will be much smaller compared to younger mine soils. A smaller $\text{Fe}^{2+}/\text{Fe}^{3+}$ ratio in mine soils would mean that Walkley-Black soil carbon values would be associated with smaller measurement errors compared to mine soils of a larger $\text{Fe}^{2+}/\text{Fe}^{3+}$ ratio (i.e., larger quantities of ferrous iron). In order to account for this changes in the chemical properties of older mine soils, coal-correction equations for Walkley-Black carbon estimates should be developed using mine soils of similar characteristics, including similar spoil type and degree of overburden weathering.

Ideally, ancillary data indicating the $\text{Fe}^{2+}/\text{Fe}^{3+}$ ratio of the mine soil of interest measured at the same point in time when the Walkley-Black carbon values were produced should be used to account for the measurement error in these carbon values induced by the ferrous iron. However, such data may not be readily available for most mine soils. Therefore, coal-correction of Walkley-Black carbon values based only on spoil type information, and developed from much younger spoil materials, may lead to over-correction of the coal error, which would lead to underestimation of the SOC present in older mine soils. Clearly, coal-corrected Walkley-Black carbon values would not meet the accuracy and precision standards of a future carbon market. However, such estimates could be used to make qualitative inferences about SOC sequestration in mine soils.

Conclusions

There were two outcomes from this project. The main outcome was a new technique for SOC content estimation on reclaimed mined lands that successfully differentiated SOC from coal-C for mine soils in the hard rock region of the Appalachian coalfield. Although this carbon partitioning technique was not expected to estimate the absolutely true SOC content in mine soils, the reported SOC estimates were carefully qualified in order to meet the accuracy and precision standards of a future global carbon market.

In this section, we presented a detailed description of a 16-step method for measuring SOC in mine soils, which was tested on a total of 30 different mine spoil mixtures representing SS- and SiS-derived mine soils in the hard-rock region of the Appalachian coalfield. We used an

acid-fumigation technique to remove the inorganic carbon from the soils (carbonates) while preserving all soluble SOC. Our methodology provided a means to correct for the carbon loss from the more volatile constituents of coal fragments in mine soils, and another correction factor for the protected organic matter in the structure of rock fragments that can remain unoxidized following thermal pretreatment.

Overall, the results of this study indicated that the mean percent relative uncertainty from the mean, PRUM (%), of SOC estimates made by a single replication of the 16-step method were lower (i.e., lower uncertainty indicating better estimation) than those for the Walkley-Black procedure for all materials except for the Jefferson forest soil. This was expected, as the Walkley-Black procedure was specifically calibrated for natural soils such as the Jefferson series. Therefore, the Walkley-Black method could be used to measure SOC in the topsoil layer of mine soils; however, the 16-step method would produce more accurate SOC values in soil layers constructed from overburden material.

The means-separation analysis for the 16-step method indicated that the 10wt%-coal samples for the SS and 50:50 SS:SiS mixtures were the only samples with significantly higher uncertainty compared to the remaining 28 mixtures that ranged in coal content from 0 to 10% by sample weight. Owing to the simplicity of the proposed SOC estimation method and its low cost of implementation, we believe that this technique will serve as a straightforward and cost-effective carbon sequestration accounting method for mine soils given that it meets the accuracy and precision standards for a future carbon market.

The uncertainties associated with the Walkley-Black procedure varied across all materials and coal levels with no trends observed. Because of their associated measurement error, mainly due to overestimation from coal and ferrous iron, some data sets of Walkley-Black soil carbon estimates for mine soils were practically useless regarding their suitability for SOC sequestration analysis on their respective research areas. Therefore, a second important outcome from this project was a coal-correction procedure, and a proposed linear regression equation, for Walkley-Black soil carbon estimates in mine soils. We developed a linear regression model ($R^2 = 55\%$) that could be used as a coal-correction formula for Walkley-Black carbon estimates on young mine soils in the Appalachian coalfield of SS, SiS, and 50:50 SS:SiS origin.

Task 1, Sub-study 2: Geospatial and Statistical Basis for Sampling Mined Land for Carbon Sequestration

Experimental

Today, existing at various stages of floristic succession – from abandoned grasslands to brushlands and forestlands – there are hundreds of thousands of hectares of land disturbed by surface mining across the United States. If reclaimed and managed properly, this land could sequester significant amounts of carbon. There are no existing mined land-specific carbon inventory guidelines. To help resolve carbon inventory and analysis issues for mined land, the objective of this work was to determine the horizontal and vertical distributions and variability of sequestered soil organic carbon (SOC), and the maximum cost-effective depth (MCD) and the minimum detectable difference (MDD) of SOC.

Data Sources

There were two sources of data for this project. The first and most comprehensive carbon data were collected from study sites in Ohio (OH), West Virginia (WV), and Virginia (VA). We established 27 treatment plots (50 x 50 m size) at each study site: (1) near the town of Jackson, Jackson County, OH; (2) near the town of Leivasy, Nicholas County, WV; and (3) near the town of Norton, VA (Figure 1-2-1). Nine plots on each site were grouped into three separate treatment blocks to allow replication of treatments. The treatment blocks were selected according to field measures of soil acidity (pH), ranging from acidic to neutral and alkaline, and site accessibility. All selected sites were on coal-mined land that had been reclaimed to grassland/hayland/pastureland in the period after 1978.

The two main criteria in choosing treatment block locations were to: (i) locate all nine plots within a block on relatively flat sites with slopes less than 8-10%; and (ii) keep the plots adjacent to each other to form a relatively contiguous matrix. These criteria were used to minimize equipment transportation between plots within a block, fencing, and other associated costs, while increasing work efficiency.

We collected surface and subsurface soil samples from shallow soil pits to 50 cm depth on each plot within a block (Figure 1-2-1), and down the soil profile to 2 m depth. We collected soil samples at 5 locations on each plot (Figure 1-2-1) to two sampling depths, 0-10 cm and 10-30 cm, using a systematic sampling approach (Dollhopf 2000). In addition to the individual soil samples, we mixed equal amounts of soil (including fine and coarse soil fragments) collected from the five shallow pits within a plot to create homogenous composite soil samples.

Two to four deep pits were excavated with a backhoe to approximately 2 m depth in representative locations at each site; these pits were excavated a few meters away from the plots to eliminate soil disturbance from soil excavation. Soil samples were collected from each soil horizon for further laboratory analysis, including total soil carbon fractionation to carbon in pedogenic (plant-sequestered) and geogenic (coal-derived) forms. Detailed descriptions of the soil profiles of each soil pit are presented in Jones (2005).

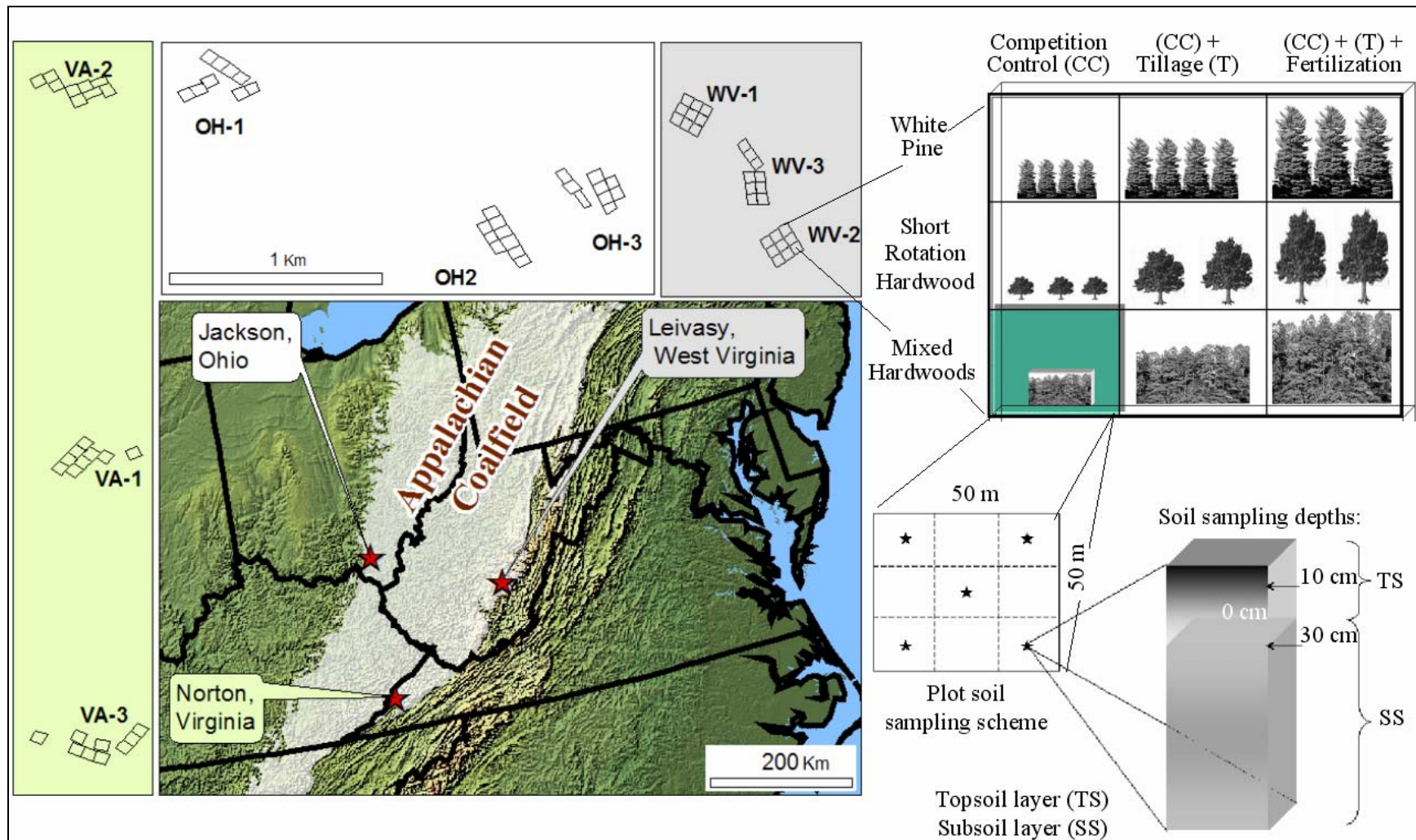


Figure 1-2-1. Study site locations, spatial distribution of treatment blocks, and data collection scheme for nine study sites located in VA, WV, and OH. The 9-cell schematic (top right corner) represents a 3 x 3 factorial experimental design (3 treatments x 3 forest types) of forest establishment treatments implemented on these sites.

All soil samples were stored in paper bags or open plastic bags in a well-ventilated area and were air-dried. The air-dried whole soil weight (Soil, g) was measured and recorded for further analysis. We passed the soil samples through a No. 10 sieve (< 2 mm) in order to separate the fine earth fraction from the coarse rock fragments. The weight of the coarse rock fragments (CFC, g) was measured and recorded after which the coarse rock fragments were discarded. The percent by sample weight of coarse rock fragments (CFC, wt%) was computed as $[\text{CFC(g)}/\text{Soil(g)}]*100$.

The fine earth component (< 2 mm) of each soil sample (fines) was used to measure soil carbon content. Percent by weight of soil organic carbon (C, wt%) contained in the fine soil fraction was determined using the 16-step SOC measurement method described in Sub-study 1. Moisture content (%) of the fines was determined as:

$$\text{Moisture (\%)} = \frac{\text{Fines(g)} - \text{Fines(g)}_{105^{\circ}\text{C}}}{\text{Fines(g)}} * 100$$

where $\text{Fines(g)}_{105^{\circ}\text{C}}$ was assumed the stable weight of the fines reached after drying the samples at 105°C for 24 hours, and Fines(g) was the weight of the fines of an air-dried soil sample.

At a randomly selected location within each 50 x 50 m plot (Figure 1-2-1), we measured the whole soil bulk density using the excavation procedure described by Blake and Hartage (1986) for a soil pit of approximately 30 x 30 cm horizontal dimensions and depth of 10 cm. The excavation hole was carefully lined with thin plastic and was filled with steel BB's to the original soil surface level in order to determine the volume of the pit. The volume of the BB's was measured in a graduated cylinder and was recorded (Jones 2005).

Following soil sample collection, nine forest establishment treatments were applied to each block (Figure 1-2-1) to test the effects of site preparation and fertilization on the survival and growth of three groups of tree species: white pine, hybrid poplar, and a mixture of native hardwoods. An extensive description of the experimental treatments and tree survival and growth data from the first growing season are presented in Casselman (2005).

Multiple bulk density samples were collected in each horizon using a modified version of the excavation method of Blake and Hartage (1986). A metal cylinder, approximately 5 cm in diameter, was used to extract soil from each horizon. The hole was lined with a thin plastic bag and filled with steel BB's to the original surface level in order to determine the volume of the excavated hole (Jones 2005). The whole soil bulk density (BD_{soil} , g cm^{-3}) was computed by dividing the whole soil weight in grams by the volume of the excavated hole measured in cm^3 units. The bulk density of the coarse rock fragments (BD_{CFC} , g cm^{-3}) was assumed 2.65 g cm^{-3} . The majority of soil minerals in the Appalachian region were assumed to have a specific gravity of 2.65 g cm^{-3} representing that of quartz and approximately the specific gravity of kaolinite (2.6 g cm^{-3}). This assumption seemed valid for most mine soils where an overburden material mostly comprised of crushed sandstone (SS) and siltstone (SiS) rock was used as a topsoil substitute material. Soils derived from SS overburden material were assumed to be comprised mostly of sand-size quartz particles, and the clay fraction of soils that are derived from SiS material was assumed to be dominated by kaolinite.

The $\text{BD}_{\text{soil}}(\text{g cm}^{-3})$, $\text{CFC}(\text{wt}\%)$, $\text{BD}_{\text{CFC}}(\text{g cm}^{-3})$, and $\text{Moisture}(\%)$ were used to determine the bulk density of the fine earth fraction (BD_{fines} , g cm^{-3}) on the oven-dry weight of the soil.

The $BD_{\text{fines}}(\text{g cm}^{-3})$ was computed as $\left[\frac{[1 - \text{CFC}(\text{wt}\%)/100]}{[1/BD_{\text{soil}}(\text{g cm}^{-3}) - \text{CFC}(\text{wt}\%)/(2.65 \times 100)]} \right] / \left[1 + \text{Moisture}(\%)/100 \right]$.

In addition, the amount of the fines as percent of the sample volume (Fines, vol%) was computed as $[100 - BD_{\text{soil}}(\text{g cm}^{-3}) * \text{CFC}(\text{wt}\%)/2.65]$. The estimated parameters for the fine earth fraction of each sample, $BD_{\text{fines}}(\text{g cm}^{-3})$ and Fines(vol%), as well as the measured C(wt%) were used to estimate the final soil organic carbon content per unit area (SOC, g m^{-2}), discussed later in the text.

The second data source was a compilation of digital remote sensing (RS) data suitable for analysis in GIS. We used the most recent aerial photography available for the area of the study sites to aid in the characterization and mapping of the sites. The exact coordinates of all soil sampling locations and plot corners on each site were overlaid on high resolution (1-m ground resolution), off-leaf condition aerial photos, available from the Virginia Base Mapping Program (VBMP 2002). Along with the high-resolution, true color imagery, the Virginia Base Mapping Program (VBMP) also provided points with elevation data, the intended use of which was to rectify the aerial imagery acquired for this program. We determined that the point-elevation data set could also be used to create digital elevation models (DEMs) for the area of the study sites. We used the *Geostatistical Analyst* extension in ArcGIS™ (ESRI Inc., Redlands, CA 92373) to create the DEMs and elevation contour data sets for the study sites.

In a preliminary GIS analysis of the VBMP point data, we determined that the vertical accuracy of a predicted DEM generated in ArcGIS 8.3 software was 2.4 m (8 ft) created at a 0.6-m (2-ft) horizontal resolution. The DEM data was used to generate aspect, slope, and curvature surface models as well as elevation contours for the extent of the study sites. The ArcGIS 8.3 software-integrated functions were used to generate all of the above surface models.

Because information about the reclamation age of the study sites was not immediately available, we performed temporal analyses of available satellite imagery (Landsat TM) for the exact location of all study sites to determine the best approximation for the month and the year when the areas of the study blocks in each study site were hydro-seeded to grass; hydro-seeding is a rapid erosion control practice commonly used in mined land reclamation that allows protective vegetation cover to be established at the spoil surface of mined sites following reclamation (Skousen and Zipper, 1996). From a soil genesis perspective, hydro-seeding could be considered the time when many biological processes begin taking place within the surface spoil layers, such as root growth and soil nutrient uptake by the vegetation and accelerated mineral weathering, slowly converting the mine spoil into a mine soil. The ground cover change on mined sites, from a bare spoil material to a grass/legume vegetation cover, can be detected from space by advanced hyperspectral imaging sensors carried by some earth observation satellites such as the Landsat 7 satellite. This rapid change in ground vegetation cover was used to determine the approximate date (+/- 4 weeks) of vegetation cover establishment on the study sites.

Data Analysis and Assumptions

All soil samples were analyzed for their carbon concentration using the 16-step SOC measurement method for mine soils described in Substudy 1. Soil organic carbon content per unit area was estimated for each soil horizon (or soil layer) with identified upper and lower

boundaries using the carbon concentration measurements (corrected for coal content and carbonates), bulk density of the Fines, the volumetric content of the Fines, and horizon depth, as depicted in the equation below:

$$\text{SOC}(\text{g m}^{-2}) = C(\text{wt}\%) * \text{BD}_{\text{fines}}(\text{g cm}^{-3}) * \text{Fines}(\text{vol}\%) * \text{Layer}(\text{cm}) \quad \text{Eq. 1}$$

where $C(\text{wt}\%)$ = percent soil organic carbon of the fine soil fraction (< 2 mm soil particles), measured on a weight basis, $\frac{C_g}{\text{Fines}_g} * 100$; $\text{BD}_{\text{fines}}(\text{g cm}^{-3})$ = bulk density of the fines measured as

$\frac{\text{Fines}_g}{\text{Fines}_{\text{cm}^3}}$; $\text{Fines}(\text{vol}\%)$ = the volumetric fraction of the fines measured as percent of the total

volume of excavated soil sample, $\frac{\text{Fines}_{\text{cm}^3}}{\text{Soil}_{\text{cm}^3}} * 100$; $\text{Layer}(\text{cm})$ = soil layer or soil horizon thickness

expressed in cm units.

Each variable in Eq.1, except $\text{Layer}(\text{cm})$, could be associated with a certain error due to the sampling and measurement techniques used. Layer thickness measurements are based on universally available standards for measuring lengths and distances. Assuming that the top and bottom boundaries of a certain soil horizon were clearly observable at a certain sampling location, the layer thickness could be measured with sufficient accuracy, e.g. ± 0.25 mm (0.5 mm), depending on the type of measurement instrument used, such as a graduated meter stick.

For this project we estimated $\text{BD}_{\text{fines}}(\text{g cm}^{-3})$ and $\text{Fines}(\text{vol}\%)$ based on the measured $\text{CFC}(\text{wt}\%)$ and $\text{BD}_{\text{soil}}(\text{g cm}^{-3})$ of the soil samples. All measurements were made for mine soil volumes of approximately $27,000 \text{ cm}^3$ ($30 \times 30 \times 30$ cm soil volume) or smaller, dependent on the maximum soil depth that could be reached by manual soil excavation. Due to the nature of most mined lands in the hard-rock region of the Appalachians, it was nearly impossible to obtain soil samples by hand beyond 30 cm. Soil sample collection to 30 cm depth was the target sampling depth for all shallow pits on our research sites. We assumed that the largest coarse rock fragments on our study sites were of equal or smaller than $27,000 \text{ cm}^3$ volume, and were contained in the excavated soil material.

Propagation of Uncertainty

Because SOC estimates are most commonly expressed in units that represent a certain area such as g m^{-2} , kg m^{-2} and Mg ha^{-1} , the results of any SOC quantification analysis would include the combined error associated with measuring the individual parameters in Eq.1. We adopted the rules of error propagation described in Harris (2005) to produce 95% confidence limits for individual $\text{SOC}(\text{g m}^{-2})$ predictions estimated at 1-cm increments down the mine soil profile for the vertical distribution of $\text{SOC}(\text{g m}^{-2})$, and the 95% confidence limits for the horizontal distribution of $\text{SOC}(\text{g m}^{-2})$ on the study sites.

Percent error for $\text{SOC}(\text{g m}^{-2})$ was estimated as $\text{SQRT}[(\% \epsilon C(\text{wt}\%))^2 + (\% \epsilon \text{BD}_{\text{fines}}(\text{g cm}^{-3}))^2 + (\% \epsilon \text{Fines}(\text{vol}\%))^2]$, where SQRT is the square-root function; the $\% \epsilon$ -term represents the percent relative uncertainty associated with sampling and measurement of the respective variables used in Eq. 1 ($C(\text{wt}\%)$, $\text{BD}_{\text{fines}}(\text{g cm}^{-3})$, and $\text{Fines}(\text{vol}\%)$). Percent relative uncertainty was estimated by dividing the absolute uncertainty, expressed in the units of the variable such as standard deviation or standard error, and the magnitude of the measurement, times 100: $\% \epsilon = [\text{StdErr}/\text{Mean}] * 100$.

Vertical Distribution and Variability of SOC in Mine Soils

Data from the deep soil pits were analyzed to determine the vertical distribution and variability of SOC down to 2 m soil depth. We used standard statistical procedures for linear regression analysis (PROC REG procedure, and the C(p) model selection method) (SAS, 2007) to model C(wt%), $BD_{\text{fines}}(\text{g cm}^{-3})$, and Fines(vol%) as a function of soil depth (independent variable). Predictions for the individual variables as a function of soil depth and their respective 95% confidence limits were also estimated and reported.

Results from the regression analyses were used to model the distribution of $\text{SOC}(\text{g m}^{-2})$ as a function of soil depth by substituting the individual predictions for C(wt%), $BD_{\text{fines}}(\text{g cm}^{-3})$, and Fines(vol%) in Eq. 1 for each 1-cm soil layer to 2 m soil depth. The vertical distribution of $\text{SOC}(\text{g m}^{-2})$ was used to determine the MDD of SOC content change on mined lands, and MCD for carbon inventorying in mine soils.

Horizontal Distribution and Variability of SOC in Mine Soils

Due to the logistics of spoil overburden material mixing, transportation, and deposition on mined sites during reclamation (Boruvka and Kozak 2001), we expected that SOC content could vary greatly across the mined landscape. In order to quantify the horizontal variability of SOC as a function of distance (i.e., determine the spatial dependence of SOC measurements (Boruvka and Kozak 2001; McKenzie and Ryan 1999)), we analyzed the $\text{SOC}(\text{g m}^{-2})$ data from the composite samples collected to 0-10 cm depth using geostatistical analyses procedures in ArcGIS™ software. We determined the auto-correlation of the data (the dependence with respect to separation distance between individual plots) using the semivariance statistic $\gamma(h)$ computed for a range of distance intervals h (Burgess and Webster 1980; Ettema and Wardle 2002; Robertson 1987) as follows:

$$\gamma(h) = \frac{1}{2N(h)} \sum_{i=1}^n [z(s_i) - z(s_i + h)]^2$$

where $N(h)$ = number of observations (i.e., plots) separated by distance h ; $z(s_i)$ = value of a certain ecosystem variable at location s_i ; and $z(s_i + h)$ = value at a location that is at distance h away from s_i . The results from the geostatistical modeling, such as the semivariogram model and maximum distance of spatial dependence, were used to determine whether these data had a spatial predictability component, and to perform kriging analysis in a GIS to create $\text{SOC}(\text{g m}^{-2})$ prediction maps across the mined landscape. For the study sites located in OH and VA, we created SOC prediction maps for the surface 0-10 cm soil depth to identify the individual plots of highest/lowest baseline SOC content.

MDD of Carbon Stocks and MCD for Carbon Inventorying on Mined Land

We used the results for the vertical distribution of SOC, as well as the results for the horizontal variation of SOC to determine the MDD of carbon stocks. We also estimated the suggested time (yr) when the next carbon inventory event should be scheduled for the sites in this study so that the measured SOC accumulation is statistically significant.

In order to determine the MCD for SOC analysis in mine soils, we estimated the proportional decrease of cumulative SOC content relative to the total amount of excavated soil

material to a certain soil depth (from 0 to 200 cm). We transformed the total amount of measured SOC to profits (monetary value) from carbon credits; the volume of excavated spoil material used for carbon analysis was transformed in terms of costs of digging a soil sample to a certain depth, including sample collection and preparation, and SOC analysis. The ratio between the profits and costs associated with SOC inventorying to a specific soil depth was used to determine the maximum cost-effective depth for SOC analysis on the study sites.

Baseline Ecosystem Carbon Content of Study Sites

One of the objectives for this research was to estimate the baseline carbon content on the study sites. Total ecosystem carbon content was computed as the sum of SOC(g m^{-2}) content to 100 cm depth, and the carbon contents in the vegetation biomass (grasses and legumes) and litter layer. We estimated the baseline SOC content on each study site using Eq. 1.

Due to the relatively small area of the mined sites in this study (approximately 6.8 ha each), and because of the observed uniformity in species composition and vigor of the vegetation within the individual study plots, which, based on our experience, is unusual for most mined sites, we sampled the vegetation biomass and litter layer of a larger mined site of similar age and land use type. We selected a mined site that was characterized with various spoil material types ranging from pure SiS to pure SS, and that was hydro-seeded to various grass and legume species. During summer 2005, we collected vegetation biomass and litter layer samples from the Flint Gap mined site located in western Russell County, VA, with respective latitude and longitude coordinates of 36°59'11"N and 82°15'20"W.

A total of 429 vegetation and litter layer samples were clipped with garden scissors from randomly chosen locations within the Flint Gap site and were stored in paper bags. The vegetation clipping area on the soil surface was marked with a square plastic frame of 0.25 m² size (0.5 x 0.5 m) for all plots except for those with large quantities of vigorously growing vegetation, on which 0.10-m² (0.2 x 0.5 m) rectangle-areas were clipped in order to fit the entire volume of clipped vegetation in the sample bag. This was necessary to make vegetation sample collection and handling more efficient. For consistency, and to increase fieldwork efficiency, we only clipped the vegetation at 5 cm above ground level, as grasses tend to form rooting clumps of various densities and sizes across the mined landscape.

The vegetation and litter layer samples were dried at 65°C, and their net weight was measured and recorded. Every 10th sample bag, one for the vegetation and another for the litter layer samples, was ground to < 1 mm size for biomass carbon content analysis; an approximately 0.300-g subsample of each ground material was used for carbon content analysis. Using the dry sample weights, carbon content measurements, and the precise sample-clipping area, we calculated the amount of carbon sequestered in vegetation and litter layer per unit area (C_{biom} , Mg ha^{-1}). We assumed that the resulting C_{biom} (Mg ha^{-1}) contents for the vegetation and litter layer carbon pools were good baseline approximations of the respective above-ground carbon pools (vegetation biomass and litter layer) for these study sites.

Results and Discussion

Study Site Characterization

The results from the temporal analyses of Landsat TM satellite imagery indicated that the best approximations for the age of the mined sites in OH were as follows: Each of the three sites was hydro-seeded to grass at the end of August (or in the beginning of September) 1994, suggesting that these sites were 9 years old at the time of soil sample collection, which occurred between July and September 2003 (Fig. 1-2-2); two of the sites (OH2 and OH3) were mined for coal in summer 1992, and the third site (OH1) was mined in spring or summer 1994, immediately after which the general areas of all three sites in OH were reclaimed to pasture/grassland post-mining land use (Figure 1-2-2).

The results for the mined sites in VA indicated that one of the study sites there (VA3) was reclaimed in June (or July) 2003, based on a Landsat satellite image of that area from June 2, 2003, suggesting that the site was bare rock and with absolutely no vegetation cover on this day; the latter suggested that the VA3 site was about 1 to 2 months old at the time of soil sampling. The other two sites in Virginia (VA1 and VA2) were reclaimed between May and October 2002; therefore, the VA1 and VA2 sites were between 1 and 1.5 years old at the time of soil sample collection (Figure 1-2-3).

Our analysis for the study sites in WV indicated that these sites (WV1, WV2, and WV3 blocks) were reclaimed between August and October 1999 suggesting that they were 4-years-old at the time of soil sample collection (Figure 1-2-4).

The soils at each plot of every site were sampled at two different depth categories (Figure 1-2-1), surface 0-10 cm and subsurface (> 10 cm depth), under the premise that the process of hydro-seeding may affect the soil chemical and/or physical properties differently at the surface layers compared to the subsurface soil layers. The average values of selected chemical and physical properties determined from analyses of the composite samples for two material types, topsoil and mine spoil, for two depth categories for each study site are presented in Table 1-2-1.

The composite-samples were constructed by mixing equivalent amounts of the whole soil sampled from the surface and subsurface soil layers at five individual locations within the 50 x 50 m plots. Only available samples from the same material type at the same depth category were mixed to create the composite samples. For those sites where topsoil was applied at the surface of the graded spoil material to a depth lower than 10 cm, composite samples were constructed for the surface topsoil layer (0-10 cm), subsurface topsoil layer (10-[X] cm in OH, and 10-[Y] cm in VA), and the underlying spoil material; the [X] and [Y] designators indicate the lower boundary of the topsoil layer in OH and VA, respectively.

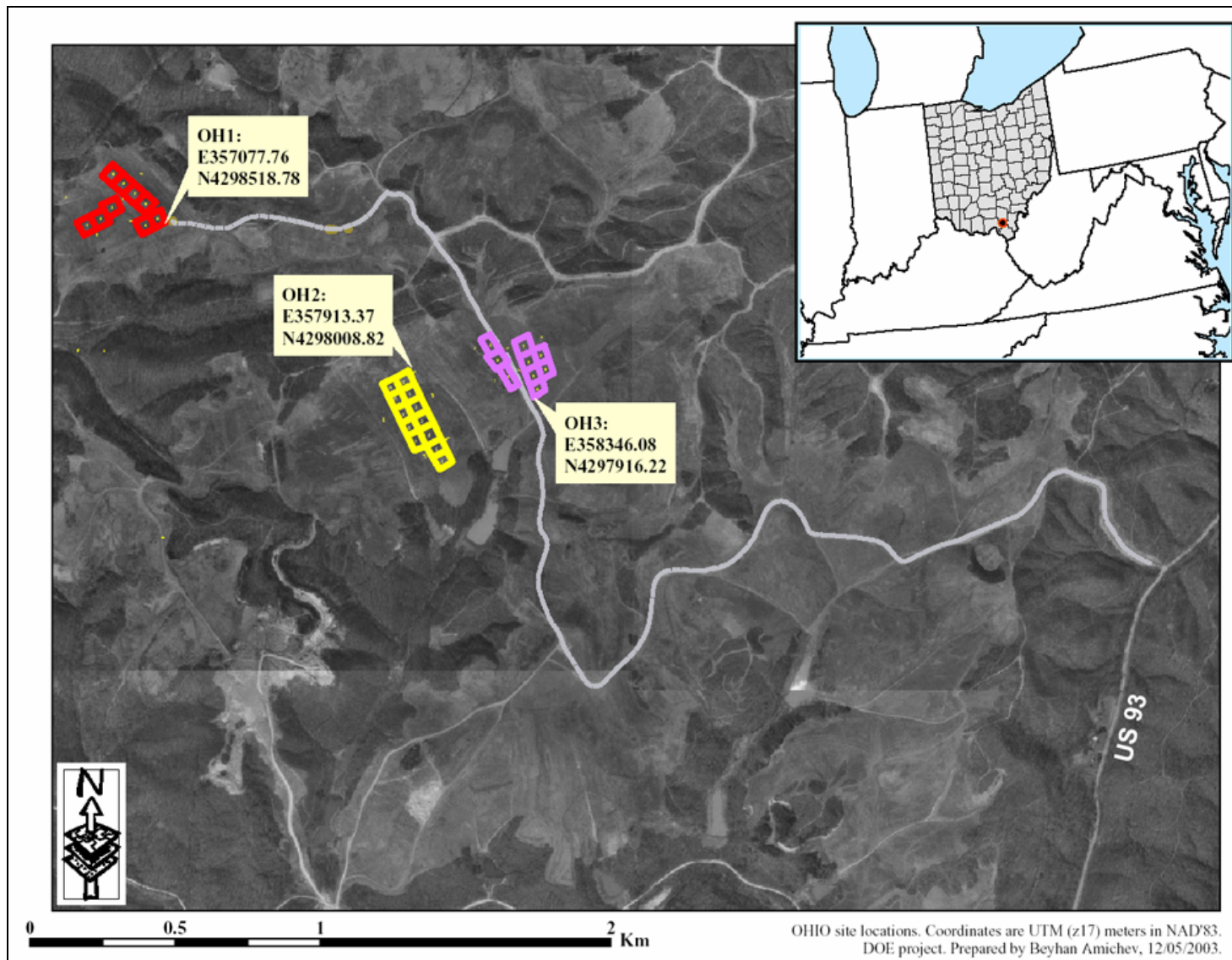


Figure 1-2-2. Location and spatial distribution of individual 50x50 m study plots located near the town of Jackson in Jackson County, OH.

VIRGINIA site locations. DOE project.
Coordinates are UTM (z17) meters in NAD'83
Prepared by Beyhan Amichev. 12/08/2003.

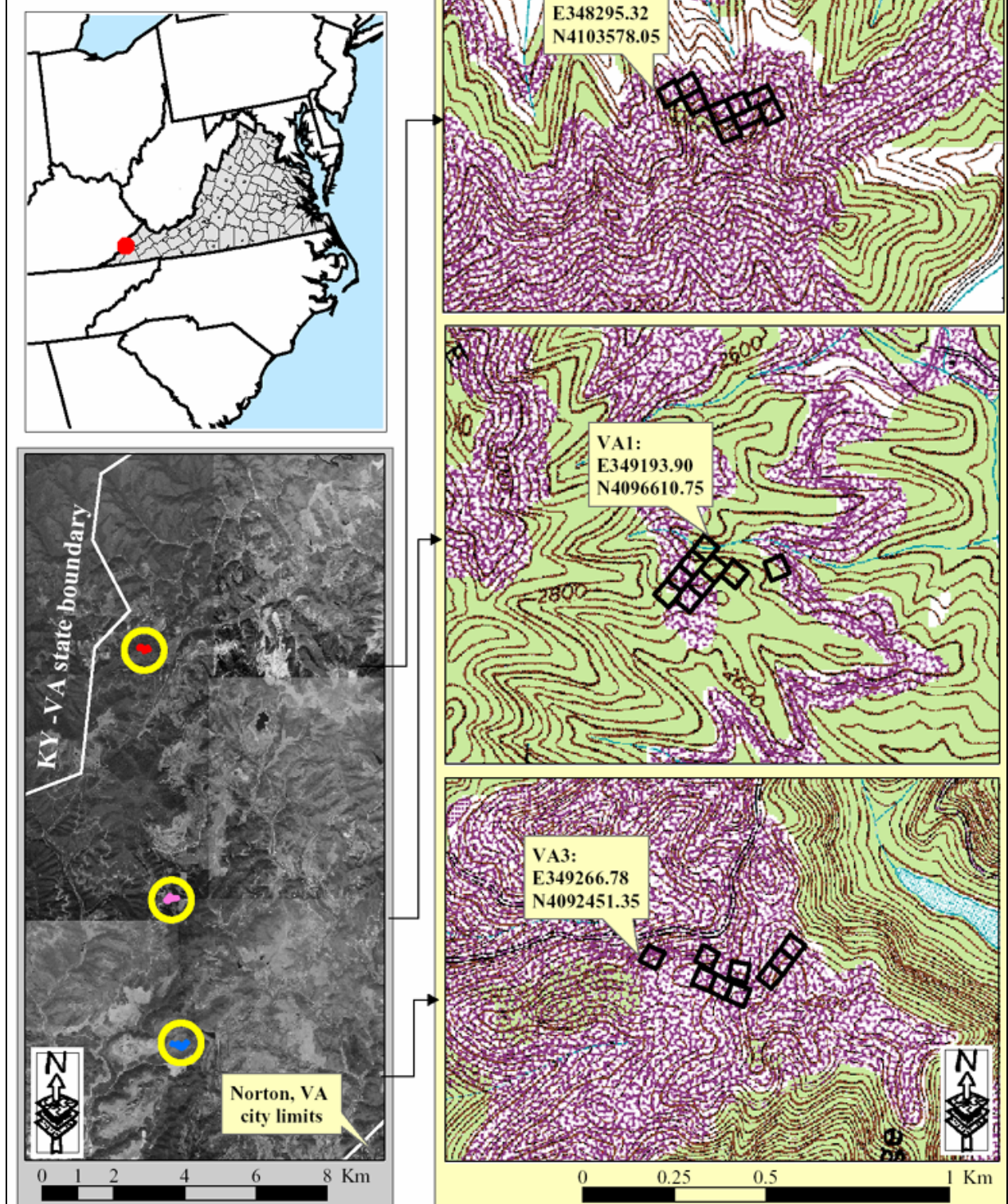


Figure 1-2-3. Location and spatial distribution of individual 50 x 50 m study plots located near the town of Norton, VA.

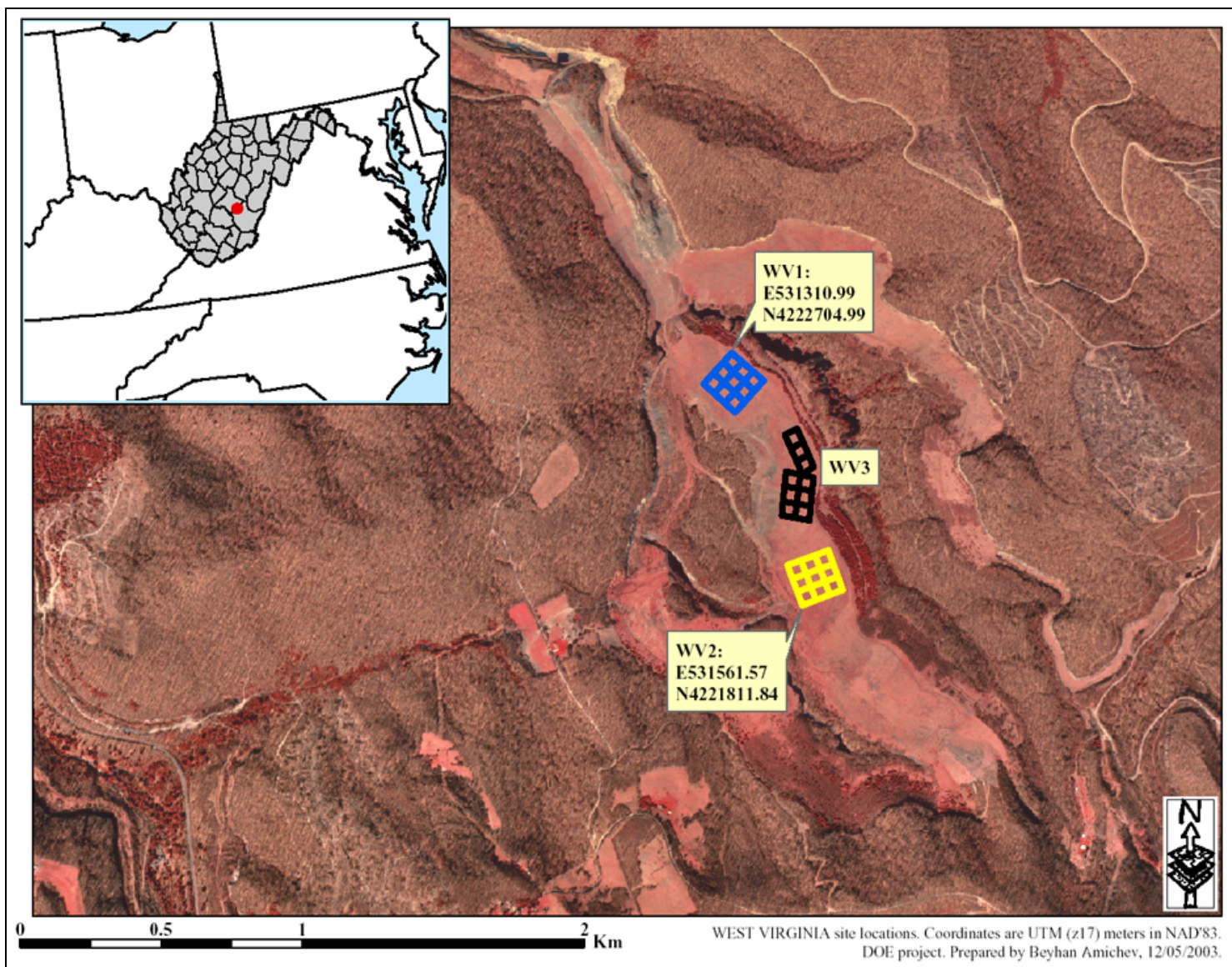


Figure 1-2-4. Location and spatial distribution of individual 50x50 m study plots located near the town of Leivasy, Nicholas County, WV.

Table 1-2-1. Average mine soil physical and chemical properties for the mined sites located in OH, VA, and WV. The results are based on analyses of composite soil samples collected between July and September 2003. Note that $BD_{\text{fines}}(\text{g cm}^{-3})$, $CFC(\text{wt}\%)$, $Fines(\text{vol}\%)$ estimates were obtained from approximately 30x30x30 cm pits excavated on each plot for the topsoil and spoil material types. The [X] and [Y] designators in the table indicate the lower boundary of the topsoil layer (> 10 cm depth) in OH and VA, respectively.

Site		OHIO				VIRGINIA				WEST VIRGINIA									
Material type		Topsoil		Spoil	Topsoil		Spoil	Topsoil		Spoil	Topsoil		Spoil						
Variable \ Depth category		0-10cm	10-[X]cm	[X]-50cm	0-10cm	10-[Y]cm	0-10cm	[Y]-30cm	0-10cm	[Y]-30cm	0-10cm	10-30cm							
<i>BD_{fines}</i>	[g cm⁻³]	1.18	<i>bd</i>	1.39	<i>a</i>	1.07	<i>cd</i>	1.24	<i>ab</i>	1.02	<i>c</i>								
	[StdDev]	0.102		0.176		0.201		0.148		0.202									
	[N]	27		17		20		12		27									
<i>Fines</i>	[vol%]	93.13	<i>a</i>	87.49	<i>a</i>	64.75	<i>b</i>	68.23	<i>b</i>	62.81	<i>b</i>								
	[StdDev]	3.378		5.914		14.101		8.343		7.111									
	[N]	27		17		20		12		27									
<i>CFC</i>	[wt%]	11.86	<i>a</i>	18.63	<i>a</i>	52.44	<i>bc</i>	46.82	<i>b</i>	57.79	<i>c</i>								
	[StdDev]	5.894		8.236		15.915		8.152		10.274									
	[N]	27		17		20		12		27									
<i>SOC_{10cm}</i>	[g m⁻²]	1,114.58	<i>a</i>	401.51	<i>adhi</i>	761.87	<i>ab</i>	94.84	<i>defgh</i>	152.81	<i>cfi</i>	301.59	<i>adi</i>	69.25	<i>cd</i>	426.53	<i>bci</i>	295.41	<i>cei</i>
	[StdDev]	398.03		123.51		488.46		133.49		202.74		612.94		152.93		650.41		373.55	
	[N]	27		3		17		20		8		7		12		27		27	
<i>C</i>	[wt%]	1.02	<i>a</i>	0.39	<i>acg</i>	0.66	<i>ac</i>	0.14	<i>deg</i>	0.20	<i>bcd</i>	0.41	<i>acd</i>	0.80	<i>ace</i>	0.63	<i>ab</i>	0.45	<i>bcd</i>
	[StdDev]	0.407		0.135		0.453		0.232		0.304		0.864		1.346		0.858		0.515	
	[N]	30		3		30		20		8		7		24		27		27	
<i>Coal</i>	[wt%]	0.49	<i>a</i>	0.00		0.10	<i>abc</i>	1.49	<i>cde</i>	1.21	<i>ade</i>	1.91	<i>bde</i>	1.75	<i>bde</i>	2.80	<i>b</i>	1.11	<i>cd</i>
	[StdDev]	0.377		0.000		0.529		0.777		0.593		0.894		1.490		1.169		0.755	
	[N]	30		3		30		20		8		7		24		27		27	
<i>pH</i>	-	5.88	<i>a</i>	6.31	<i>ab</i>	6.84	<i>bc</i>	5.99	<i>a</i>	5.93	<i>a</i>	6.38	<i>acde</i>	6.73	<i>be</i>	6.12	<i>a</i>	6.66	<i>bd</i>
	[StdDev]	0.486		0.610		0.608		0.566		0.687		0.366		0.625		0.299		0.515	
	[N]	27		6		27		25		17		10		27		27		27	
<i>EC</i>	[mmho cm⁻¹]	0.10	<i>a</i>	0.37	<i>bc</i>	0.47	<i>b</i>	0.26	<i>cde</i>	0.36	<i>bd</i>	0.34	<i>bce</i>	0.28	<i>cde</i>	0.21	<i>ace</i>	0.11	<i>a</i>
	[StdDev]	0.051		0.398		0.288		0.123		0.161		0.127		0.098		0.052		0.02	
	[N]	27		6		27		25		17		10		27		27		27	
<i>SS</i>	[wt%]	22.96	<i>b</i>	5.00	<i>b</i>	16.48	<i>b</i>	63.00	<i>ac</i>	83.89	<i>a</i>	61.25	<i>ad</i>	54.58	<i>cd</i>	9.07	<i>b</i>	9.81	<i>b</i>
	[StdDev]	21.629		8.660		15.493		31.058		18.671		27.613		37.239		6.207		7.401	
	[N]	27		3		27		25		9		8		24		27		27	
<i>Sand</i>	[wt%]	41.89	<i>a</i>	33.00	<i>ab</i>	28.63	<i>b</i>	53.20	<i>c</i>	56.00	<i>ce</i>	56.00	<i>cf</i>	66.50	<i>def</i>	70.00	<i>d</i>	61.19	<i>cd</i>
	[StdDev]	6.333		0.00		12.762		13.404		14.000		14.967		9.459		0.000		11.744	
	[N]	27		3		27		25		9		8		24		27		27	
<i>Topsoil</i>	[cm]			19.91			<i>a</i>			17.03					<i>a</i>			-	
	[StdDev]			6.646						12.201								-	
	[N]			27						27								-	

*Estimates followed by the same letter are **not** significantly different ($P>0.05$) by Tukey's HSD means-comparison procedure. .

Among other soil properties, such as those presented in Table 1-2-1, the most apparent difference between the topsoil and spoil materials for all sampling locations was their color; beige-brown for topsoil, and gray for the overburden spoil material. The different coloring of these materials is a direct result of the difference in their “weathering stage” as well as the difference in organic matter content. The original topsoil, often thousands of years old, was bulldozed, stored, and re-applied on the graded mined site in OH and VA. During physical and chemical weathering processes, the less stable primary soil minerals break down to form more stable secondary minerals of different oxidation state. For example, the gray-colored material in the spoil is mostly due to the presence of reduced iron II (Fe^{2+}) which, after weathering, becomes part of different secondary soil minerals such as Fe-(oxy)hydroxides, in which the iron exists in its oxidized state, iron III (Fe^{3+}). Secondary Fe-containing soil minerals range in color from yellow-orange to reddish-brown. The darker color of the topsoil material compared to the spoil could also be due to the accumulation of plant-sequestered soil organic matter. Because of all of the above reasons, the distinction between the topsoil and spoil material in the field was very apparent during sample collection.

The foremost difference between the study sites in the three states was that these mine soils were constructed very differently. All three sites in OH had topsoil placed on top of the spoil surface so that topsoil was present at each sampling location. The average topsoil depth in OH was about 20 cm. At certain sampling locations in OH, the extreme thickness of the topsoil (50+ cm deep in some locations) prohibited acquisition of the spoil material underneath; extreme efforts were required to remove the deep topsoil layer in order to reach the spoil material underneath (Table 1-2-1). Some areas of the VA study sites were also reclaimed with topsoil placed at the surface; approximately 65% of all individually sampled locations had topsoil cover (Table 1-2-1). The average topsoil depth on the sites in VA was 17 cm. The mine soils in WV did not have any topsoil cover.

The mine soils in VA, both topsoil and spoil materials, had significantly higher SS content compared to the respective materials in OH and WV. While the mine soils in OH were a mixture of SS and SiS, the mine sites in WV had developed mostly from dark shale-derived overburden material. It should be noted that for the C (wt%) analysis of the soil samples from WV, the dominant mine soil material was assumed to be SiS to match the estimation procedures developed for the 16-step SOC estimation method (Substudy 1). This assumption seemed valid because both shale and SiS spoil materials weather to finer-textured mine soils, thus complying with the assumptions used for the 16-step SOC estimation method.

The pH results for all sites suggested that the soil acidity levels ranged between slightly acidic (pH 5.88) to almost neutral (pH 6.84) (Table 1-2-1). In all instances, except for VA, the pH of the subsurface spoil material was significantly higher compared to the surface topsoil material. These results were as expected, because soil carbonates (Ca -, Na -, and Mg - CO_3) from the surface soil layers were most likely dissolved and leached out from the system. Our results indicated that the soil acidity levels of the surface topsoil and spoil materials in the three states were not significantly different at the 0.05-alpha level (Table 1-2-1).

Although there were no observed root-growth restrictions due to elevated soil-salt content, there was a clear trend of salt accumulation as a function of depth for the OH sites (Table 1-2-1). The study sites in OH were the oldest among all sites (at age 9) indicating that for nine growing seasons approximately 270% and 370% higher amount of salts accumulated in the subsurface topsoil and spoil layers, most likely due to leaching from upper soil layers, relative to the surface

topsoil layer. The salt content was higher in the surface spoil layers in VA and WV compared to the spoil material beneath in the respective states, although these differences were not statistically significant (Table 1-2-1).

The values for BD_{fines} (g cm^{-3}), Fines(vol%), and the CFC(wt%) for the topsoil and spoil materials presented in Table 1-2-1 were measured from one excavated pit of approximately 30 x 30 cm horizontal dimensions, at representative locations within each plot. These measurements were used to estimate the soil organic carbon content for a 10-cm-thick soil layer ($\text{SOC}_{10\text{cm}}$) within the surface (0-10 cm) and subsurface (>10 cm depth) layers of the topsoil and spoil materials within the respective plots.

The highest $\text{SOC}_{10\text{cm}}$ content for the surface layers was observed on the sites in OH ($1,115 \text{ g m}^{-2}$), which was statistically higher than the $\text{SOC}_{10\text{cm}}$ on the sites in WV (427 g m^{-2}) and VA (95 and 302 g m^{-2} , respectively, for surface topsoil and spoil materials) (Table 1-2-1). On average, the $\text{SOC}_{10\text{cm}}$ content ranged between 69 and $1,115 \text{ g m}^{-2}$ in the following decreasing order (by soil material, depth, and state): $\text{Topsoil}_{(0-10), \text{OH}} > \text{Spoil}_{(>10), \text{OH}} > \text{Spoil}_{(0-10), \text{WV}} > \text{Topsoil}_{(>10), \text{OH}} > \text{Spoil}_{(0-10), \text{VA}} > \text{Spoil}_{(>10), \text{WV}} > \text{Topsoil}_{(>10), \text{VA}} > \text{Topsoil}_{(0-10), \text{VA}} > \text{Spoil}_{(>10), \text{VA}}$ (Table II-1). A similar trend was observed for the C(wt%) concentrations measured in these soils.

Based on the $\text{SOC}_{10\text{cm}}$ estimates in the surface soil layers for the sites in OH and WV, we estimated the average carbon sequestration rates expected on these sites at $124 \text{ g C m}^{-2} \text{ yr}^{-1}$ in OH, and $107 \text{ g C m}^{-2} \text{ yr}^{-1}$ in WV. However, these estimates would be based on the assumption that at the time of grass sowing on the respective sites, the SOC content was zero, and SOC accumulation was only due to plant growth (Table 1-2-1). However, as mentioned earlier, in addition to the coal fragments in the spoil materials for the sites in this study, the shale- and SiS-derived spoils could contain significant amounts of old carbon (organic matter incorporated in soil rock structures during sediment deposition and rock formation), which could have the same properties as recent, plant-derived SOC. It would only be possible to produce valid SOC sequestration rates at a future carbon inventory for these sites, when the present $\text{SOC}(\text{g m}^{-2})$ content (i.e., baseline value) would be subtracted from the respective future $\text{SOC}(\text{g m}^{-2})$ values, and the difference would be divided by the number of years between the two carbon inventories.

The coal content was highest in WV ranging from 1.1 to 2.8wt%, followed by the soils in VA (from 1.2 to 1.9 wt%), and OH (from 0 to 0.5wt%). The coal-derived carbon content was greater than the pedogenic carbon content in all sites, except OH (Table 1-2-1). Regardless of material type and sampling depth, the coal-derived carbon ranged from 118 to 964% greater than plant-derived carbon content in the sites in VA, and ranged from 147 to 344% in the sites in WV (Table 1-2-1). As mined sites get older and additional carbon is incorporated in the soil, the content of coal-derived carbon is expected to become smaller relative to the increasing content of plant-derived C.

Unless coal-derived carbon is accounted for during SOC analysis using mine-soil-specific estimation methods such as the 16-step method, the amount of plant-sequestered carbon in mine soils could be significantly overestimated. Dependent upon the quality of coal fragments in certain mine soils, due to microbial degradation, the coal-derived carbon could also be incorporated into the biogeochemical cycle of carbon in mine soils (Hayashi et al. 1999; Rumpel and Kogel-Knabner 2002; Waschkies and Huttl 1999). However, the microbial degradation of coal could take hundreds to thousands of years to completely transform the resistant coal

fragments to bio-available carbon forms; therefore, accurate and precise quantification of coal in mine soils is necessary for carbon inventories on mined land.

The $BD_{\text{fines}}(\text{g cm}^{-3})$ was found to be significantly higher in the spoil materials compared to the topsoil layer in OH and VA (Table 1-2-1). The spoil material in WV had the lowest $BD_{\text{fines}}(\text{g cm}^{-3})$ among all spoil materials analyzed. The latter could be due to the fact that shale could have lower specific gravity relative to SS and SiS materials, largely due to the higher organic matter content that is incorporated in most shale materials during sedimentation and rock-forming processes millennia ago. However, during sample collection, we encountered equal difficulty of spoil excavation on all study sites suggesting that all spoil materials were equally-compacted due to the heavy grading by mining equipment during the reclamation process. Overall, the bulk density of the fines ranged from 1.02 to 1.39 g cm^{-3} , and decreased in the following order: Spoil_{OH} > Spoil_{VA} > Topsoil_{OH} > Topsoil_{VA} > Spoil_{WV} (Table 1-2-1).

The highest bulk density of the spoil materials in OH was associated with significantly lower CFC(wt%) compared to the spoil materials in VA and WV (Table 1-2-1). Although there was no observable trend of this for the rest of the data, there may be a logical explanation of the relationship between low CFC(wt%) and high $BD_{\text{fines}}(\text{g cm}^{-3})$ measurements on mined land. On one hand, it is not unusual for soils with low rock content to have lower structural stability and to bed down, thus becoming more compacted and with higher bulk density of the fines compared to the more stable rocky soils. On the other hand, rocky soils (i.e., soils with higher CFC(wt%) content) have less soil material in the < 2-mm soil fraction per unit of soil volume; hence, the estimate for their $BD_{\text{fines}}(\text{g cm}^{-3})$ is low. Fine-textured soils such as clays could become very compacted with time, leading to root growth limitations relative to coarser-textured sandy soils, in which root growth could be virtually unlimited due to their low soil bulk density, provided there is sufficient source of soil nutrients from fertilization or other sources.

The content of the Fines(vol%) (93 and 87vol% for the topsoil and spoil materials, respectively) on the sites in OH was significantly higher compared to the sites in VA and WV. In contrast, the CFC(wt%) content (12 and 19 wt% for the topsoil and spoil materials, respectively) on the sites in OH was significantly lower compared to the sites in VA and WV. This is a well observed trend in young mine soils where coarse rock fragments weather rapidly to sand- and silt-sized soil particles thus leading to a reduction in CFC(wt%) content in the soil, and contributing to a higher content of the fines (Bendfeldt et al., 2001; Haering et al., 1993). As a result of the increase of the fines, the potential of mine soils to store SOC(g m^{-2}) is expected to increase due to greater soil mineral surface area where SOM absorption processes could occur more readily. As mine soils develop, larger quantities of sequestered SOC in mine soils could become more resistant to chemical degradation from microbial activity, thus, transforming the mine soil environment into a significant carbon sink.

For most rocky and compacted soils such as mine soils, digging deeper and wider soil pits and sampling large volumes of soil for a more complete soil analysis, and to achieve accurate measurements of CFC(wt%) and $BD_{\text{soil}}(\text{g cm}^{-3})$ across the mined landscape can be very costly and laborious. Currently, measurement of CFC(wt%) in mine soils poses the greatest challenge for carbon inventorying on mined land when the goal is to produce accurate and precise SOC(g m^{-2}) estimates across a certain project area. Because of the anthropogenic origin of mine soils, large boulders, often as large as an average-size golf cart, could be randomly distributed below the soil surface across the mined landscape. It is apparent that in order to account for the rock

fragments of various sizes in mine soils, one must sample soil volumes that are big enough to contain the largest of soil rock fragments present.

However, often constrained by time and available resources, most mined land researchers collect soil samples of less than $27,000 \text{ cm}^3$ volume (approximately 1 ft^3), which is the volume of a soil pit with $30 \times 30 \times 30 \text{ cm}$ horizontal and vertical dimensions. That is, the results from such studies are often based on the assumption that the largest rock fragments on these mined lands are of volumes smaller than $27,000 \text{ cm}^3$. However, failure to test this assumption on a certain mined site could infer that most of the soil volume is comprised of fines, which would lead to significant overestimation of SOC (g m^{-2}) evident from the relationship depicted in Eq. 1.

Mine soils have coarse rock fragment contents ranging from 40-80% by soil weight. The size of the coarse rock fragments and their spatial distribution is quite different from those of undisturbed soils. The main reason for this is that the underlying layers of unconsolidated rock were blasted and put on the reclaimed sites at random during surface coal mining operations. Therefore, some parts of the project area may end up with larger coarse rock fragments that could be of different rock type than others. Unfortunately there is no established research approach for accurately predicting the distribution, size, and content of mine soil rock fragments.

The following hypothetical example shows the significant effect of underestimation of CFC on SOC inventorying in mine soils. Assume a soil pit (Pit 1) dug precisely to $1 \times 1 \times 1 \text{ m}$ dimensions using a backhoe (Figure 1-2-5). The excavated soil was separated into its fine-earth ($< 2 \text{ mm}$) and coarse rock fragment ($> 2 \text{ mm}$) fractions. The fine earth soil fraction was used for further analyses to measure $C(\text{wt}\%)$, $BD_{\text{fines}}(\text{g cm}^{-3})$, and $\text{Fines}(\text{vol}\%)$ soil properties. The carbon concentration was measured at $2.3 \text{ wt}\%$; bulk density of the fines was estimated at 1.1 g cm^{-3} , and the volume of the fines was 60.8% . The volume of the coarse rock fragments was measured to be exactly equal to the volume of six identical, spherically-shaped boulders 0.5 m in diameter (Figure 1-2-5a). By substituting the above measured parameter values in Eq. 1, the resulting SOC estimate to 1 m soil depth was computed at $15,382.4 \text{ g m}^{-2}$.

Now consider an alternative sampling scenario for SOC estimation on the same site at a second sampling location (Pit 2) (Figure 1-2-5b). Assume a 0.5-m -deep hand-dug pit of $1 \times 1 \text{ m}$ horizontal dimensions. On this pit, because of time and funding constraints, we assumed that the mine soil of the lower half-meter of the profile ($0.5\text{-}1.0 \text{ m}$ soil layer) had the exact same physical and chemical properties as that of the upper half-meter ($0\text{-}0.5 \text{ m}$ layer). Although this assumption seemed valid for Pit 1, in the case for Pit 2, it did not hold true (Figure 1-2-5b). Despite the fact that Pit 1 and Pit 2 were located on the same site, the vertical distribution of coarse rock fragments down the soil profile was clearly different; more than 75% of the coarse rock fragments in Pit 2 were located in the $0.5\text{-}1 \text{ m}$ soil layer, while the distribution was about 50% in the upper- and lower-halves of Pit 1 (Figure 1-2-5b).

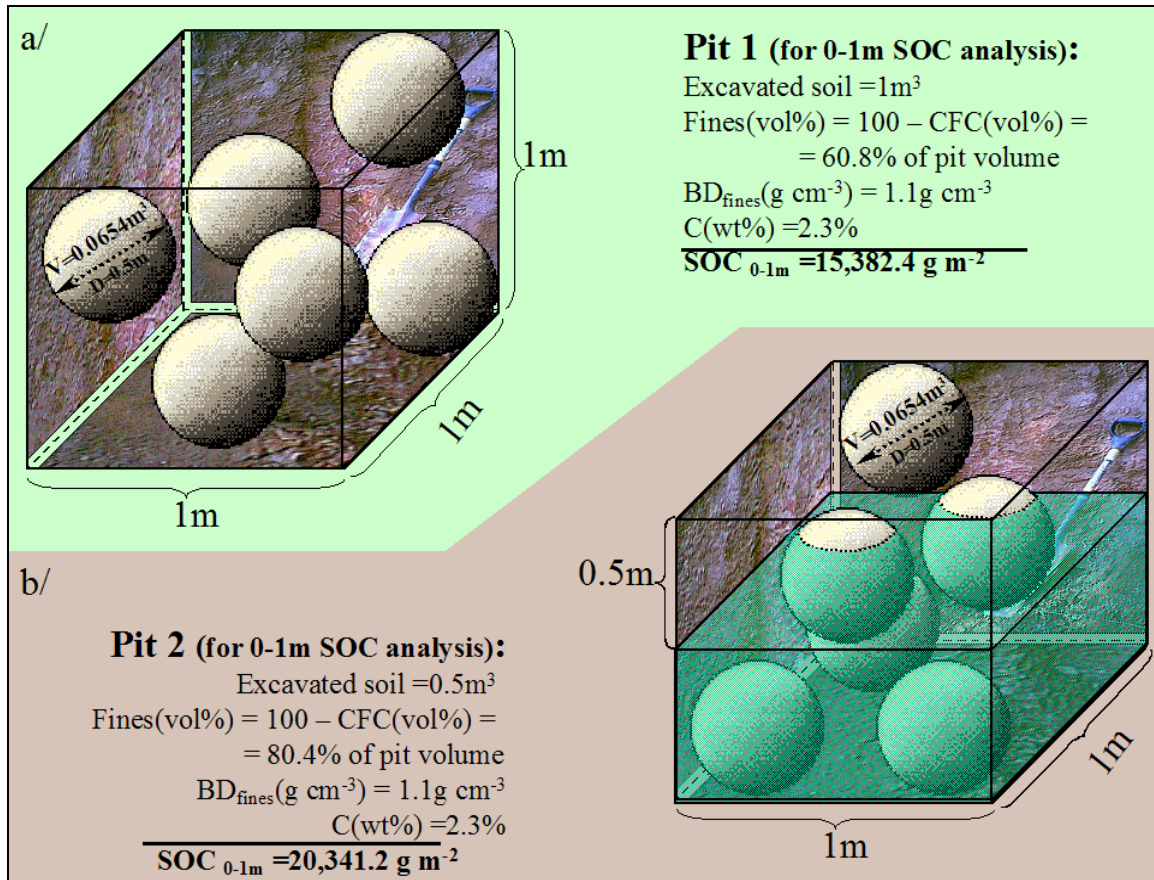


Figure 1-2-5. Hypothetical example for the combined effect of soil sampling volume and coarse rock fragment content on the accuracy of SOC estimation from two adjacent mine soil pits. Note that the difference between the two CFC(vol%) measurements is due to the depth of soil sampling.

Using similar analyses, we estimated the carbon content for Pit 2 at 2.3 wt%, bulk density of the fines equal to 1.1 g cm^{-3} , and volume of the fines equal to 80.4%. Related to the volume of the fines, due to the shallower depth of Pit 2 and the assumptions that the upper- and lower-halves of Pit 2 are similar, we failed to account for approximately 50% of the total volume of coarse rock fragments located in the lower half-meter of the profile (Figure 1-2-5b). By substituting the parameter values measured for Pit 2 in Eq.1, the carbon content to 1 m soil depth was estimated at $20,341.2\text{ g m}^{-2}$ ($= 203.4\text{ Mg ha}^{-1}$). As a result, the carbon content measured in Pit 2 was 32% greater compared to Pit 1 ($15,382.4\text{ g m}^{-2} = 153.8\text{ Mg ha}^{-1}$), which was due to an inadequate measurement of the coarse rock fragment content in Pit 2.

The example shown in Figure 1-2-5 demonstrated the magnitude of $\text{SOC}(\text{g m}^{-2})$ overestimation on mined land that could occur due to sampling error in the measurement of CFC content in mine soils. This clearly indicates that more research is needed to develop practical and inexpensive methods for CFC measurement in mine soils that would help produce accurate SOC estimates for carbon credits. Because rock weathering could be a very slow process, CFC determination can be assumed a one-time operation for a certain mined site, the results of which could be used for future carbon sequestration inventories during the entire extent of the project (e.g. 60 yrs).

Based on our findings in Substudy 1 regarding accuracy of the 16 step SOC estimation method, the mean of the measured C(wt%) in mine soils was within 16% to 23% of the true SOC. The latter was demonstrated to hold true for mine soils developed from weathered SS or SiS overburden spoil-materials of up to 10wt% coal fragment content. Similar evaluation studies for the $BD_{\text{fines}}(\text{g cm}^{-3})$ and Fines(vol%) variables could potentially help determine the remaining sources of sampling and measurement error associated with the final product from Eq.1, the $SOC(\text{g m}^{-2})$ estimate.

The forest site quality class (FSQC) of all study sites in OH, VA, and WV was estimated and reported by Jones (2005); FSQC range from class I to V, I being the best. The three sites in OH were classified as FSQC II with soil depth considered most limiting soil property for reforestation and tree root growth (Jones 2005). The 1-year-old sites in VA were also classified as FSQC II, and the 0.2-year-old site was of FSQC III. Jones (2005) reported that the root growth limiting properties for the VA sites were insufficient soil depth and high soil density. The WV sites were classified as FSQC III with main limiting factors being soil density and limited soil depth. In his discussion, Jones (2005) cautioned that the reported FSQC classifications may not be appropriate for the OH sites due to the lack of subsurface drainage data that could affect the current FSQC estimates for these sites. Jones (2005) also cautioned the use of the FSQC estimates for the VA study sites where the soil properties of these young mine soils were expected to change with time which could lead to the development of mine soils of significantly different FSQC.

For anthropogenic soils such as mine soils, the spatial distribution of soil properties was shown to reflect specific human activities during site reclamation such as spoil mixing and transportation, instead of natural soil development processes (Boruvka and Kozak, 2001). Because of the intensity of coal mining operations, overburden spoil materials from different geologic strata are simultaneously deposited on previously mined land that is targeted for reclamation and revegetation. The resulting mined landscape is usually a vast field of a mixture of overburden material types, which begin to exhibit distinctly different chemical and physical properties shortly after vegetation cover is established.

Due to the interactions between young mine soils and the vegetation which they support, very distinctive patterns of vegetation vigor, and varieties of vegetation species can be observed on most mined lands. For example, based on our reconnaissance on various mined sites, some legume species such as clover (*Trifolium* spp.) appeared to form dense stands on mine soils of pH greater than 6-6.5 and grew poorly under more acidic soil conditions. In comparison, the more aggressive warm-season legume species such as sericea lespedeza (*Lespedeza cuneata* (Dum.-Cours.) G. Don) grew equally well on a wide range of mine soil acidity. As a result of the selective behavior and the competition between the different vegetation species in mine soils, one could observe inconsistent patterns of vegetation distributed across the mined landscape, often detecting islands of completely different species of grass and legumes within a field of homogenous grass/legume vegetation of another species. Therefore, the SOC concentrations could be expected to differ across the mined landscape as a function of spoil material type on which the C-sequestering vegetation had developed (Boruvka and Kozak, 2001).

Vertical Distribution and Variability of SOC in Mine Soil

It is not intuitive for the common person to always picture the top-most portion of the earth's crust to have variable depth, and be characterized with different chemical and physical properties that could vary both horizontally across the landscape and vertically down the soil profile. Soils are three-dimensional landscape features that are constantly changing and are affected by many factors. The type of vegetation growing on a certain soil, the underlying geology, climate, topography, and time (think thousands of years) are all factors that have led to the formation of one type of soil or another across the globe. The implications of all this to carbon sequestration studies in terrestrial ecosystems is that researchers must account for each of the above factors affecting the properties of soils in order to accurately model and predict the amount of sequestered carbon in the soil.

An alternative approach to determine the carbon sequestration potential of different soils is to be able to accurately and precisely measure the carbon that is currently sequestered in a certain soil in the horizontal and vertical dimensions. Both approaches for predicting sequestered carbon in soils are valid and are widely used among researchers (Akala and Lal 2001b; Amichev et al. 2004; Parton et al. 1987). However, empirical models, which are based on measured data, are usually regarded superior to process-based models. On one hand, process-based modeling relies on a researcher's complete and thorough understanding, and the comprehensive and valid modeling of all processes occurring in soils that could affect the amount of sequestered carbon for different soils. On the other hand, empirical models are associated with higher costs. In order to determine the amount of carbon sequestration down the soil profile, many soil samples must be excavated to a desired depth, sampled, and analyzed for carbon in a cost-effective way so that future carbon sequestration studies could be warranted.

Presented in Figures 1-2-6 to 1-2-11 are the vertical distributions of the chemical and physical properties used for SOC(g m^{-2}) analysis on the study sites in OH, VA, and WV. The three variables in Eq. 1, Fines(vol%), $\text{BD}_{\text{fines}}(\text{g cm}^{-3})$, and C(wt%), were modeled as a function of soil depth (independent variable) using standard linear regression analysis techniques. These analyses were performed using the deep-pit soil data for the respective sites in OH, VA, and WV.

All regression models for the sites in OH were significant: content of fines ($P = 0.1254$, Figure 1-2-6a); bulk density of the fines ($P = 0.0694$, Figure 1-2-6b); and C(wt%) ($P = 0.0009$, Figure 1-2-7a). The respective R^2 values of the regression models for the Fines(vol%), $\text{BD}_{\text{fines}}(\text{g cm}^{-3})$, and C(wt%) variables were 0.18, 0.24, and 0.28. Similarly, we developed regression models for the distribution of these three variables down the mine soil profiles for the sites in VA (Figures 1-2-8a, 1-2-8b, and 1-2-9a); the respective R^2 values of these models were 0.60 (Fines(vol%)), 0.52 ($\text{BD}_{\text{fines}}(\text{g cm}^{-3})$), and 0.34 (C(wt%)). Due to the variation of the data for the WV sites, we were unable to develop prediction models that could be used to describe SOC(g m^{-2}) down the mine soil profile for these sites (Figures 1-2-10 and 1-2-11); as the WV sites age and greater amounts of SOC accumulate in the soil, the variation of SOC may decrease, thus allowing to develop the above-mentioned prediction models as a function of soil depth.

In OH, the regression models indicated that the content of the fines was between 70% and 97% by soil volume, the bulk density of the fines was between 0.7 and 1.7 g cm^{-3} , and the C(wt%) was between 0% and 1.9% (Figures 1-2-6 and 1-2-7). Soil organic carbon concentration for the sites in OH decreased with the increase of soil depth following a reciprocal function by depth [$\text{C(wt\%)} = f(1/\text{Depth})$]. However, the variation among individual C(wt%) measurements

did not permit conclusive inferences about this soil property. The 95% confidence limits for C(wt%) indicated that, at the lower limit of the predictions, there would not be any organic carbon accumulation in the soil below 3 cm depth; however, at the upper limit of the model predictions, the results showed that there would be at least 1.376 wt% of organic carbon at 200 cm depth (Figure 1-2-7a). The upper limit predictions for C(wt%) could be regarded as a computational error assuming that root growth on grasslands is usually concentrated in the surface soil layers. Because of the excessive compaction of mine soils, root growth is generally restricted to the surface 1 m of the mine soil (Haering et al., 1993).

However, it could be argued that SOC accumulation in subsurface soil horizons at depths greater than 1 m could be due to the translocation of dissolved soil organic matter down the profile (Jansen et al. 2003), and its adsorption to the surfaces of fine soil particles located in the voids between larger coarse rock fragments. An alternative explanation for the relatively high SOC content at lower soil depths could be the presence of old organic matter. At soil depths beyond the expected rooting zone, significant amounts of old organic matter deposited during rock formation could be contained within the layers of sedimentary rocks such as shale and SiS; the latter (along with SS) are common rock types that overlay coal seam in the Appalachian coalfield, and are used for mined land reclamation as a topsoil substitute material (Daniels et al. 2004).

In order to test the hypothesis that the measured C (wt%) at soil depths beyond the maximum observed rooting depth (assume soil depth >1 m) is due to translocation of dissolved SOM, we recommend measuring the C(wt%) content at the same depth of the same mined site after a certain period of time long enough to allow significant detectable SOC additions to occur. Only if significant additions were observed in the future, then for this site there would be sufficient evidence to support the tested hypothesis.

Because the 16-step SOC estimation method (Substudy 1) was specifically designed to partition total soil carbon into carbon from coal and carbon from plant material (i.e., recent carbon), the C(wt%) results from this analysis were corrected for coal content in the soils, but there was no correction applied for the old organic carbon contained within the organic fractions embedded in the shale and SiS rock fragments. In practice, regarding SOC(g m⁻²) inventories for carbon credits on mined land, any measurement error due to the presence of old carbon in shale- and SiS-derived spoil materials will be eliminated after subtracting the baseline SOC(g m⁻²) estimates from the SOC(g m⁻²) values obtained at a future carbon inventory. This would hold true for most mined sites for which the overburden material used during reclamation is a homogenous mixture of blasted rock, or better, within the boundaries of a delineated homogenous stratum on a certain project site, the overburden material used for reclamation is of the same spoil type. In contrast, coal fragments are often randomly distributed across the mined landscape and down the soil profile, and should always be accounted for in SOC(g m⁻²) analyses on mined lands.

Using the individual predictions for C(wt%), Fines(vol%), and BD_{finest}(g cm⁻³) (Figures 1-2-6 and 1-2-7a) in Eq. 1, we estimated the SOC content at 1-cm increments down to 200 cm soil depth. The 95% confidence limits of the SOC(g m⁻²) estimates were computed using the rules of error propagation described above, and the resulting predictions were evaluated against the SOC of both deep-pit and composite-soil samples (from shallow-pits), for which all three parameters were measured at the same location within the soil profile; the SOC(g m⁻²) of the composite-samples was estimated using an average sampling depth of the five shallow pits excavated within each plot.

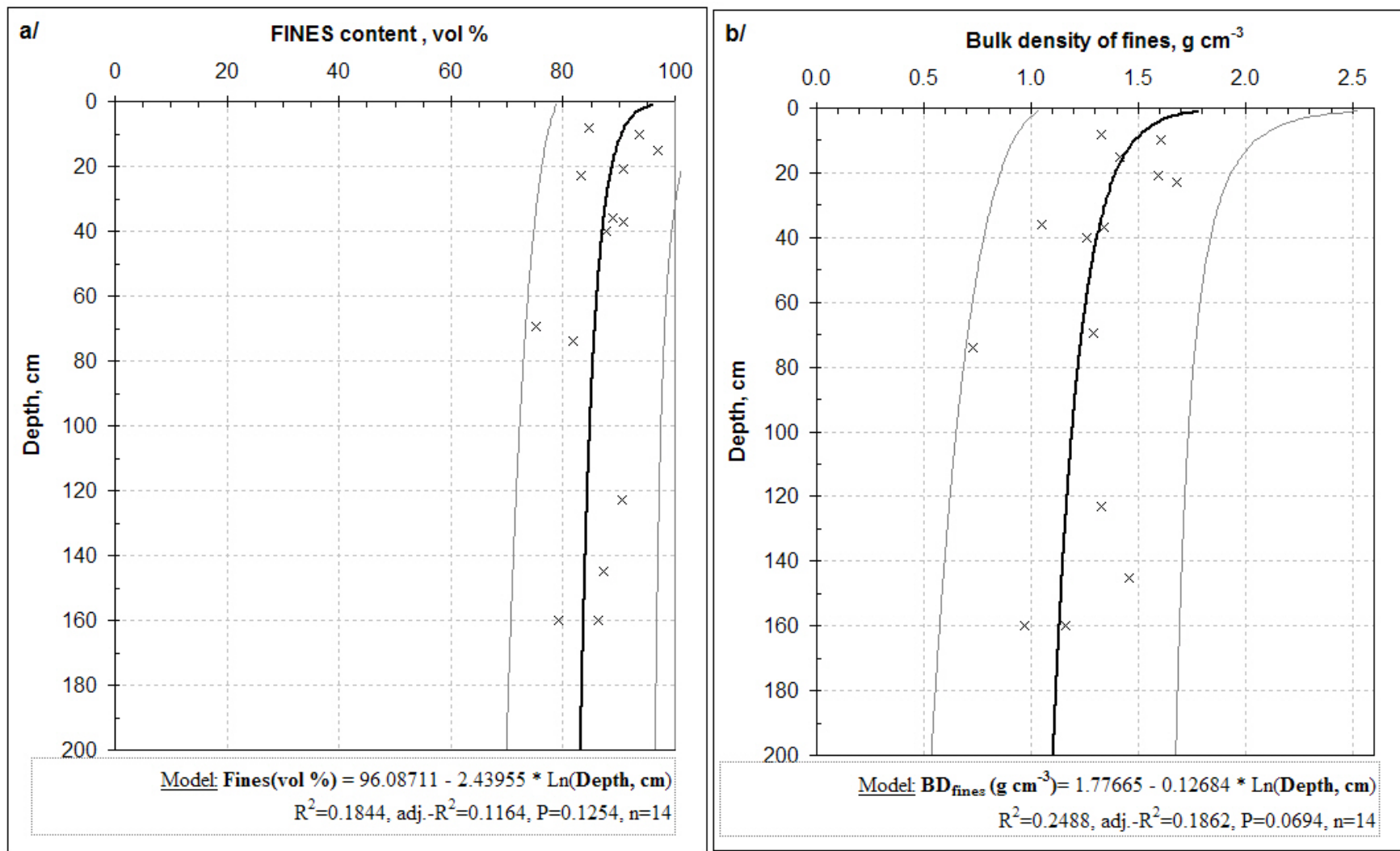


Figure 1-2-6. Vertical distribution of the content of the fine soil fraction, expressed as percent by soil volume (a), and bulk density of the fines (b) down the soil profile of 8 deep soil pits excavated on study sites located in OH. Darker line indicates the mean prediction, and lighter lines indicate the 95% confidence limits of these predictions, made by the depicted regression models.

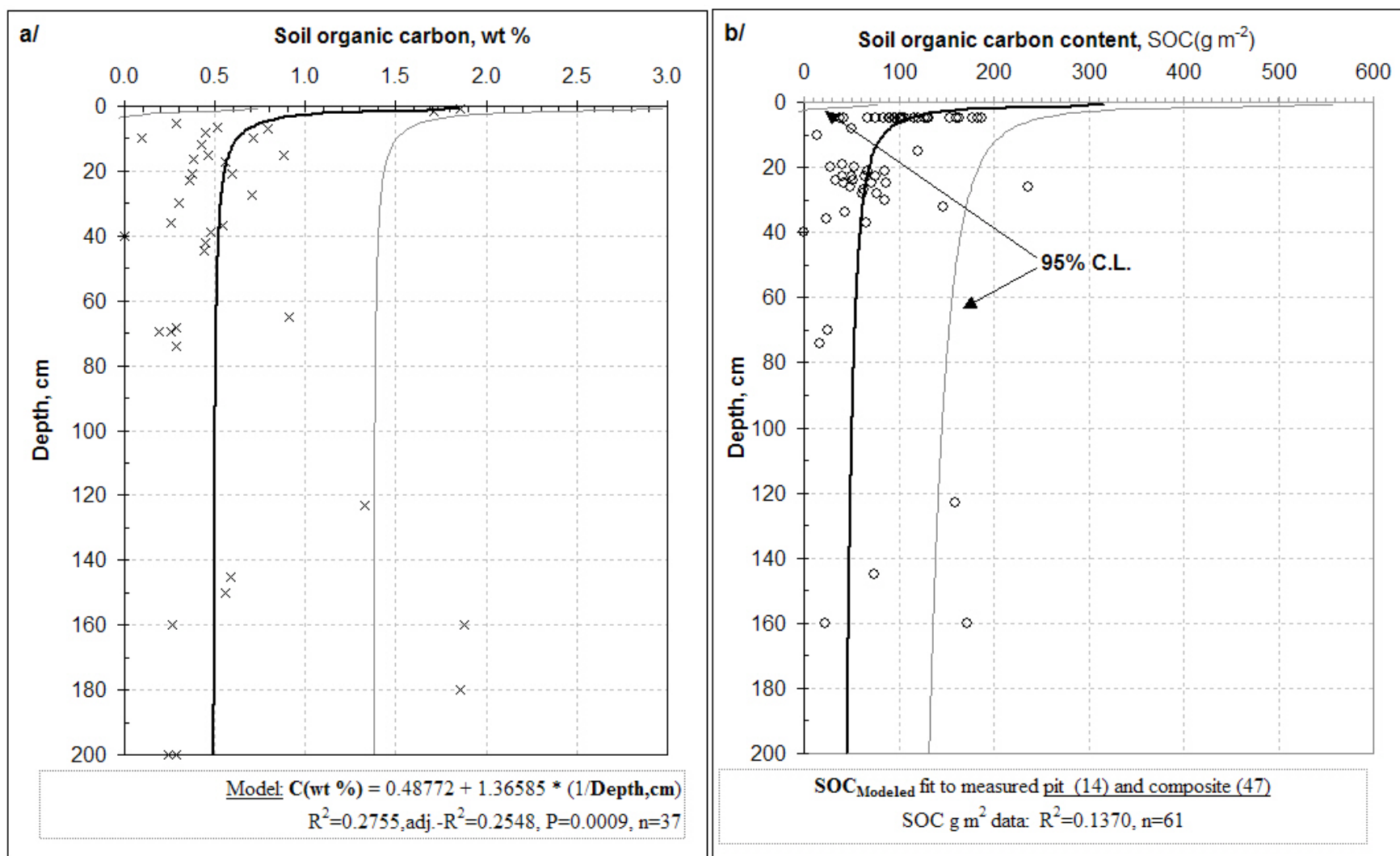


Figure 1-2-7. Vertical distribution of C(wt%)(a) and the modeled SOC(g m⁻²) content (b), which was computed in Eq.1 based on data collected from 8 deep soil pits excavated on study sites located in OH; 95% C.L. indicates the confidence limits of the mean SOC estimate at the 95% probability level.

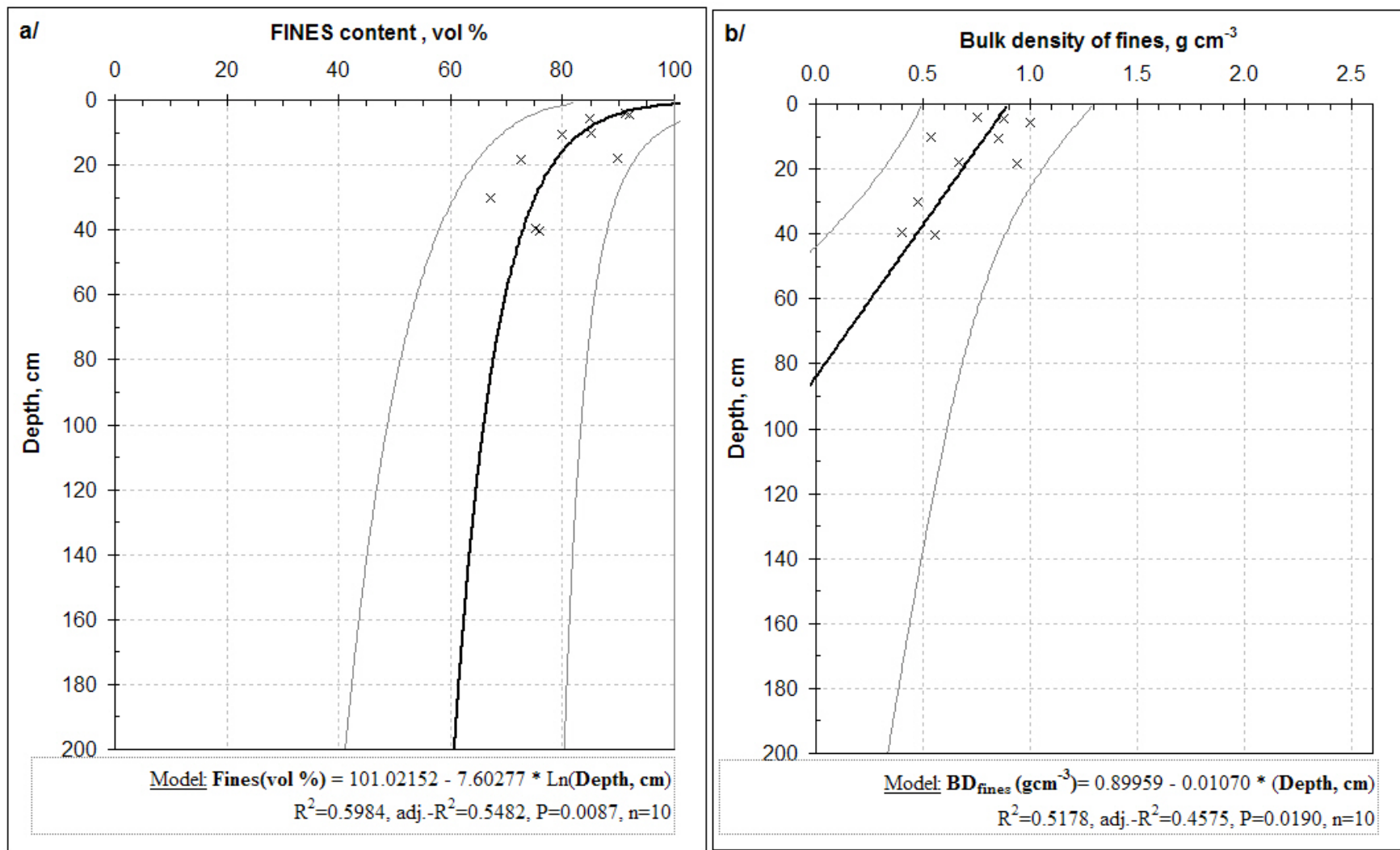


Figure 1-2-8. Vertical distribution of the content of the fine soil fraction, expressed as percent by soil volume (a), and bulk density of the fines (b) down the soil profile of 6 deep soil pits excavated on study sites located in VA. Darker line indicates the mean prediction, and lighter lines indicate the 95% confidence limits of these predictions, made by the depicted regression models.

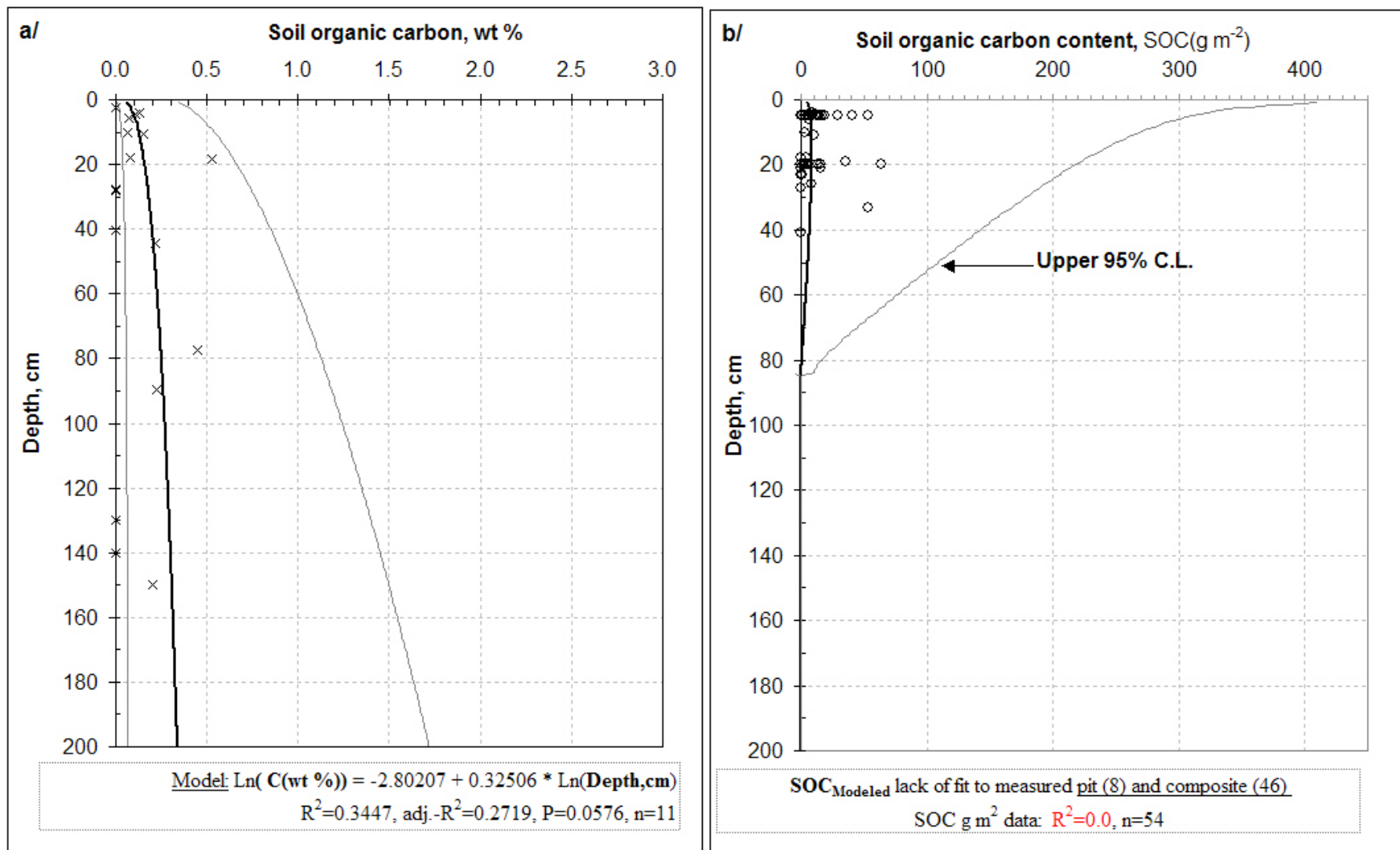


Figure 1-2-9. Vertical distribution of C(wt%)(a) and the modeled SOC(g m⁻²) content (b), which was computed in Eq. 1, down the soil profile of six deep soil pits excavated on study sites located in VA; 95% C.L. indicates the confidence limits of the mean SOC estimate at the 95% probability level.

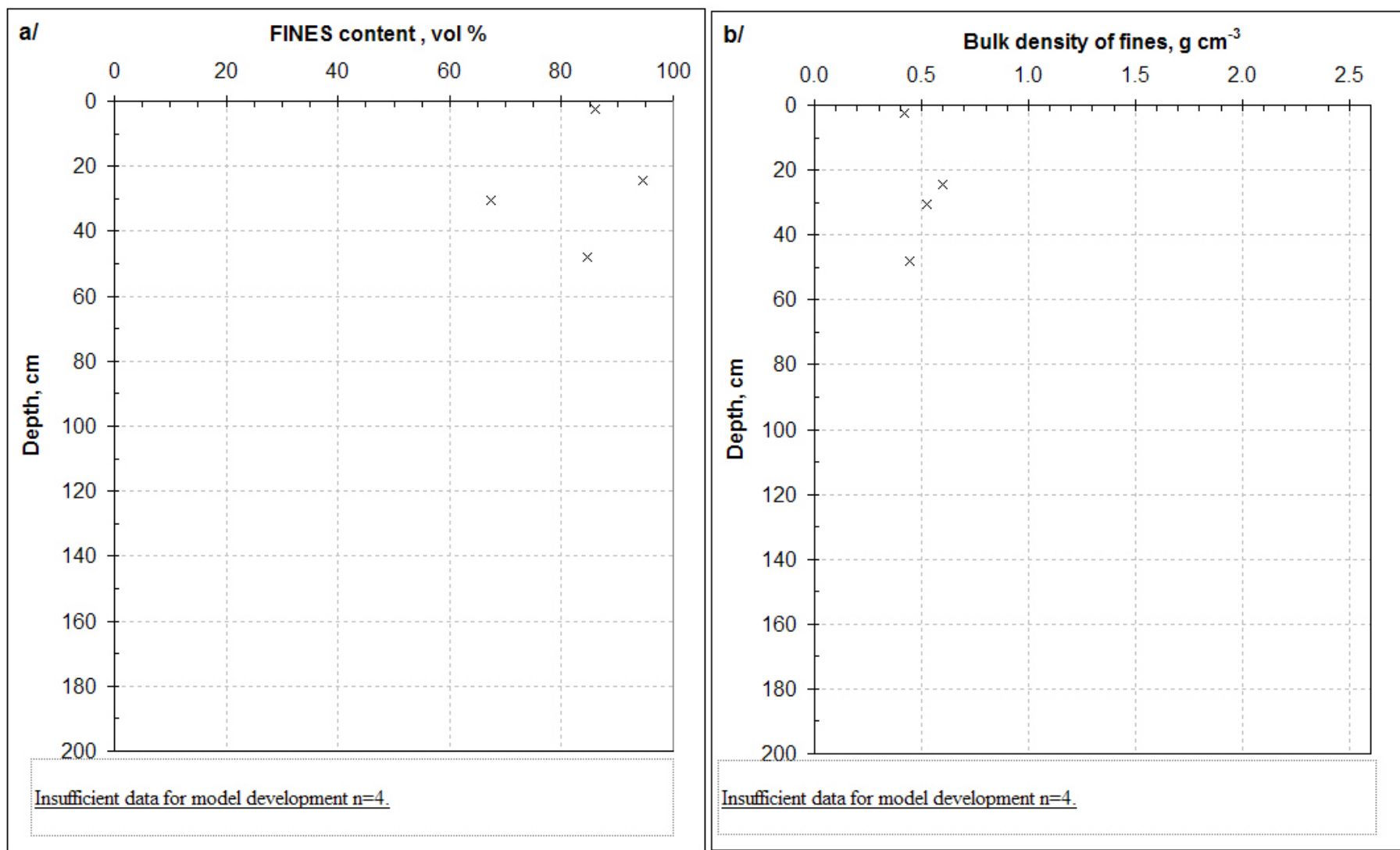


Figure 1-2-10. Vertical distribution of the content of the fine soil fraction, expressed as percent by soil volume (a), and bulk density of the fines (b) down the soil profile of three deep soil pits excavated on study sites located in WV.

In OH, the individual SOC(g m^{-2}) measurements within the entire soil profile ranged between 0 and 237 g C m^{-2} ; we used these estimates to determine the goodness-of-fit of the SOC(g m^{-2}) prediction model, which could be used as the best approximation of the vertical SOC distribution on the study sites in OH (Figure 1-2-7b). Based on the goodness-of-fit analysis, the R^2 of the SOC(g m^{-2}) model was 14%. That is, 86% of the variation of sequestered SOC down the mine soil profile in OH could be explained by factors or soil properties other than soil depth, or that additional data is required in order to encapsulate the total variation of SOC down to 200 cm soil depth.

The SOC model for the study sites in OH suggested that the average SOC(g m^{-2}) content was between 316 g m^{-2} (at 1 cm depth) and 45 g m^{-2} (at 200 cm depth) (Figure 1-2-7b). We used this SOC(g m^{-2}) model to determine the MCD and MDD for carbon inventorying on these sites, which is described in later sections of this text; both estimates of MCD and MDD are critical pieces of information for a cost-effective carbon sequestration analysis on mined lands.

In VA, the content of the fines ranged between 67% and 93% by soil volume, the bulk density of the fines ranged between 0.4 and 1.0 g cm^{-3} , and SOC content ranged between 0 and 0.5% by soil weight (Figures 1-2-8 and 1-2-9). The individual SOC estimates down the soil profile ranged between 0 and 63 g C m^{-2} . Despite the relatively high R^2 values of the prediction models for the three input variables in Eq.1 (ranging from 34% to 60%, Figures 1-2-8 and 1-2-9a), the final model for predicting SOC (g m^{-2}) content by depth had poor fit based on the validation results with the data for the composite and deep-pit soil samples in VA (Figure 1-2-9b).

One possible explanation for the poor fit of the SOC prediction model for the sites in VA could be the young age of these soils (~ 1 yr). Despite the fact that the content of the fines and the bulk density of the fines were predicted with relatively good regression models, due to the low carbon levels in these soils and their relatively high variability by depth, the final SOC(g m^{-2}) prediction model was of poor fit. Our results indicated that in one growing season very little carbon was added to mine soils, evident from the observed low SOC(g m^{-2}) amounts within a 1-cm-thick soil layer ($\sim 10 \text{ g m}^{-2}$) from the composite and deep-pit soil samples in VA. In contrast, the 9-year-old sites in OH ($\sim 87 \text{ g m}^{-2}$) contained an average of 770% more SOC within a 1-cm-thick soil layer (per 1 m^2 horizontal area) at any depth down the soil profile.

Due to the high variability of the data for the study sites in WV, we did not develop regression models that could describe the amount of sequestered carbon down the mine soil profile in these sites. The data showed that the content of the fines ranged between 67% and 95% by soil volume, the bulk density of the fines ranged between 0.4 and 0.6 g cm^{-3} , and the SOC content ranged between 0 and 1.5% by soil weight (Figures 1-2-10 and 1-2-11). The individual SOC content for a 1-cm layer down the profile ranged between 0 and 137 g m^{-2} .

The results in this study indicated that young mine sites have low SOC content which is characterized with a relatively high variability in the vertical dimensions. The direct implications to carbon sequestration studies on mined land would be that it may be practically impossible to model SOC(g m^{-2}) as a function of depth for sites younger than nine years. Therefore, it could be impractical and very costly to inventory the amount of sequestered SOC in young mine sites (< 9 yrs), as the variability in the data would not allow the estimation of the site-specific MCD and MDD parameters (described in later sections), which are necessary for cost-effective carbon inventories on mined land.

Horizontal Distribution and Variability of SOC in Mine Soil

The horizontal distribution of SOC(g m⁻²) across the mined landscape was determined from the SOC content of the composite samples collected at each plot (Figure 1-2-1). Because the mined landscape is a two-dimensional plane, as opposed to the one-dimensional soil depth, different modeling approaches other than regression models must be used in order to describe the horizontal distribution of SOC(g m⁻²).

Assuming that all soil samples from the study sites were collected independent of each other, and were analyzed to produce average SOC(g m⁻²) estimates, then these results could be used as a surrogate measure of the horizontal distribution of SOC(g m⁻²) (Table 1-2-2). Although results for the mean SOC(g m⁻²) content and the associated standard error of the mean are the most likely estimates to be reported for carbon credits, these values contain only a fraction of the information needed to describe the horizontal distribution of sequestered carbon on mined lands. That is, by producing average SOC(g m⁻²) estimates, a very important piece of information is omitted, the spatial location of the sequestered carbon and the spatial dependence between sampling locations that are separated by a certain distance. Additionally, if SOC(g m⁻²) estimates of two sampling locations were dependent upon each other, then some assumptions for statistical analysis may be violated (Robertson, 1987).

Table 1-2-2. Soil organic carbon results for the surface 0-10 cm depth of nine study sites located in OH, WV, and VA. The results are based on analyses of composite soil samples collected from each 50 x 50 m plot in the three states. Values in parenthesis are standard error of the respective mean estimates.

STATE	Site	N	SOC _{0-10cm} g m ⁻²	SOC _{0-10cm} ----- wt % -----	Coal_C _{0-10cm}
OH	OH1	9	1,254.25 (110.91) ^a	1.070 (0.091)	0.533 (0.106)
	OH2	9	838.57 (122.42) ^b	0.772 (0.142)	0.679 (0.156)
	OH3	9	1,250.90 (125.05) ^a	1.238 (0.128)	0.217 (0.073)
	Average	27	1,114.58 (76.60) ^A	1.027 (0.078)	0.476 (0.075)
WV	WV1	9	152.26 (68.48) ^c	0.232 (0.099)	3.481 (0.217)
	WV2	9	526.11 (165.70) ^c	0.948 (0.300)	2.180 (0.380)
	WV3	9	601.21 (325.82) ^c	0.707 (0.364)	2.750 (0.443)
	Average	27	426.53 (125.17) ^C	0.629 (0.165)	2.804 (0.225)
VA	VA1	9	115.64 (28.56) ^d	0.146 (0.036)	0.973 (0.106)
	VA2	9	59.78 (59.49) ^d	0.113 (0.111)	2.188 (0.188)
	VA3	9	269.91 (178.25) ^d	0.376 (0.251)	1.632 (0.316)
	Average	27	148.44 (63.32) ^D	0.212 (0.091)	1.598 (0.156)

* SOC(g m⁻²) values within a state followed by the same letter are **not** significantly different at the 0.05 alpha level (PROC GLM in (SAS®), Tukey's HSD means comparison)

Therefore, a modeling approach with spatial capabilities must be used to predict the amount of sequestered carbon across the landscape that would successfully account for any existing spatial dependence between sampling locations. Furthermore, spatial modeling

techniques such as the semivariogram analysis could allow the estimation of SOC(g m^{-2}) for areas between sampling locations (via interpolation) across the mined landscape, which would be of much greater practical use than a single mean estimate.

There are some advantages to reporting mean SOC(g m^{-2}) estimates for a certain mined site (Table 1-2-2). The two inferences that could be made from average estimates and their associated variation (estimate of the standard error of the mean) regarding the carbon sequestration potential of a certain mined site are: (i) the relative ease of comparison of sequestered carbon between two specific project sites; and (ii) the relatively straight-forward estimation of SOC variability at each site.

Our results indicated that the average amount of sequestered carbon in the 0-10 cm layer in the three sites in OH was 161% greater than the carbon amount sequestered in WV, and 653% greater than the carbon amount sequestered in VA (Table 1-2-2). Statistical analysis for mean comparison between the sites of each state showed that the average amount of sequestered carbon in the surface 0-10 cm layer was significantly smaller in the OH2 site compared to the other two sites in OH. There was no evidence that the average SOC(g m^{-2}) estimates per treatment block for the sites in WV ($P=0.4735$) and VA ($P=0.2779$) were significantly different from the other two treatment blocks within the respective states (Table 1-2-2).

The coefficient of variation ($\text{CV}\% = \text{Std.Dev} / \text{Mean} * 100$) of sequestered carbon for the three OH sites was between 27 and 44%, and between 94 and 163% for the WV sites, and between 74 and 299% for the VA sites. On one hand, the observed differences in CV% estimates could be the result of the different ages of the respective mine sites, so that, as the sites aged, distinct soil horizons developed with discernible better soil structure (Haering et al. 1993), making older mine soils more homogeneous compared to younger sites. On the other hand, as sites aged, higher amounts of carbon could be sequestered in the soil, the cumulative amount of which could become much greater than the background variation of SOC(g m^{-2}) across the landscape.

Widely available, there are statistical tools for analysis of spatially distributed data. Semivariogram analysis is the technique that is commonly used to determine the spatial dependence of a certain soil property across the landscape (Boruvka and Kozak 2001; Ettema and Wardle 2002; McKenzie and Ryan 1999; Saldana et al. 1998; Zhang and McGrath 2004). Spatial models could be developed from any spatial input data set and could be produced by using automated procedures in ArcGIS® spatial analysis software (geostatistical wizard), or in SAS® statistical software (PROC VARIOGRAM procedure). Provided there were sufficient data to develop a suitable spatial model, evident from the semivariogram analysis, predictions for SOC(g m^{-2}), and the estimation error associated with these predictions could be computed for all unsampled areas located between any pair of samples. Also, using ordinary kriging procedures (Boruvka and Kozak 2001; Burgess and Webster 1980; Robertson 1987), the resulting spatial models could be used to create maps of the horizontal distribution of SOC across the landscape; SOC maps could be of great value and use for planning future carbon inventories, and for making important land management decisions.

We used the surface SOC(g m^{-2}) data for the 0-10 cm depth for the three sites in OH and WV to determine whether sequestered carbon in mine soils was spatially dependent, that is, to determine whether the SOC for two sampling locations that were spatially close to one other would have more similar SOC estimates compared to sampling locations that were further apart.

Ultimately, we determined whether there was a distance between two sampling locations in our carbon data sets, beyond which SOC measurements were no longer similar. This site-specific distance is often referred as the *range* of heterogeneity, beyond which SOC estimates of two sampling locations are independent of one other (Ettema and Wardle, 2002).

We assigned the SOC(g m⁻²) content of the composite soil samples as the SOC(g m⁻²) at the center location for each 50 x 50 m plot of the respective plots in OH and WV. To achieve normal distribution of the data, and to comply with the equal variance assumption for geostatistical analysis, prior to spatial modeling, the SOC estimates in both data sets were transformed using the natural-log transformation [SOC' = ln(SOC)]. The resulting semivariograms, and the SOC prediction maps for the sites in OH and WV, are depicted in Figures 1-2-12 and 1-2-13, respectively. The spatial analysis of the combined data for all VA sites did not result in a suitable semivariogram model, largely because of the great separation distance between the individual research blocks.

The results indicated that SOC(g m⁻²) values for the surface 0-10 cm soil layer were spatially dependent on the sites in OH and WV with estimated range distances of 576 and 593 m, respectively (Figures 1-2-12 and 1-2-13). Based on the ratio between the spatially-dependent variance (Ettema and Wardle, 2002), and the total variance of the data, the SOC(g m⁻²) for the sites in WV had better spatial structure (i.e., higher spatially-dependent predictability) compared to the sites in OH. The spatial structure for the sites in OH was estimated at 0.439 (0.088999/(0.088999+0.11385)), and 0.764 (2.8955/(2.8955+0.89619)) for the sites in WV (Figures 1-2-12 and 1-2-13).

The nugget variance, the value of the semivariance at 0-m separation distance, indicated that approximately 56 and 24% of the total variation in SOC(g m⁻²) for the 0-10 cm depth for the sites in OH and WV, respectively, could not be explained by the spatial models depicted in Figures 1-2-12 and 1-2-13. In order to improve these models, additional data collected at higher sampling intensities than the current sampling intensity of four samples per hectare (estimated as nine composite samples per 2.25-ha study area) would be required for the sites in both states. An achievable goal would be to collect sufficient data to improve these spatial models to a spatial structure greater than 0.80, which would allow the development of more accurate SOC(g m⁻²) prediction maps.

Because of the limited number of soils samples and the relatively large separation distances between sampling locations in OH and WV, the spatial structures of the respective SOC data sets could be affected by the block allocation of the samples. A very similar block effect was observed in the SOC data used in the geostatistical analysis for the OH sites. The surface SOC measurements for the OH2 site, in particular, were significantly lower than the estimates in the other two blocks in OH (Table 1-2-2, Figure 1-2-12). In order to eliminate any potential block effects in future geostatistical analyses of SOC data, additional samples could be collected between the blocks. Ideally, the data set should be based on samples that are collected from systematically distributed locations across the entire extent of the mined site without any gaps between the blocks.

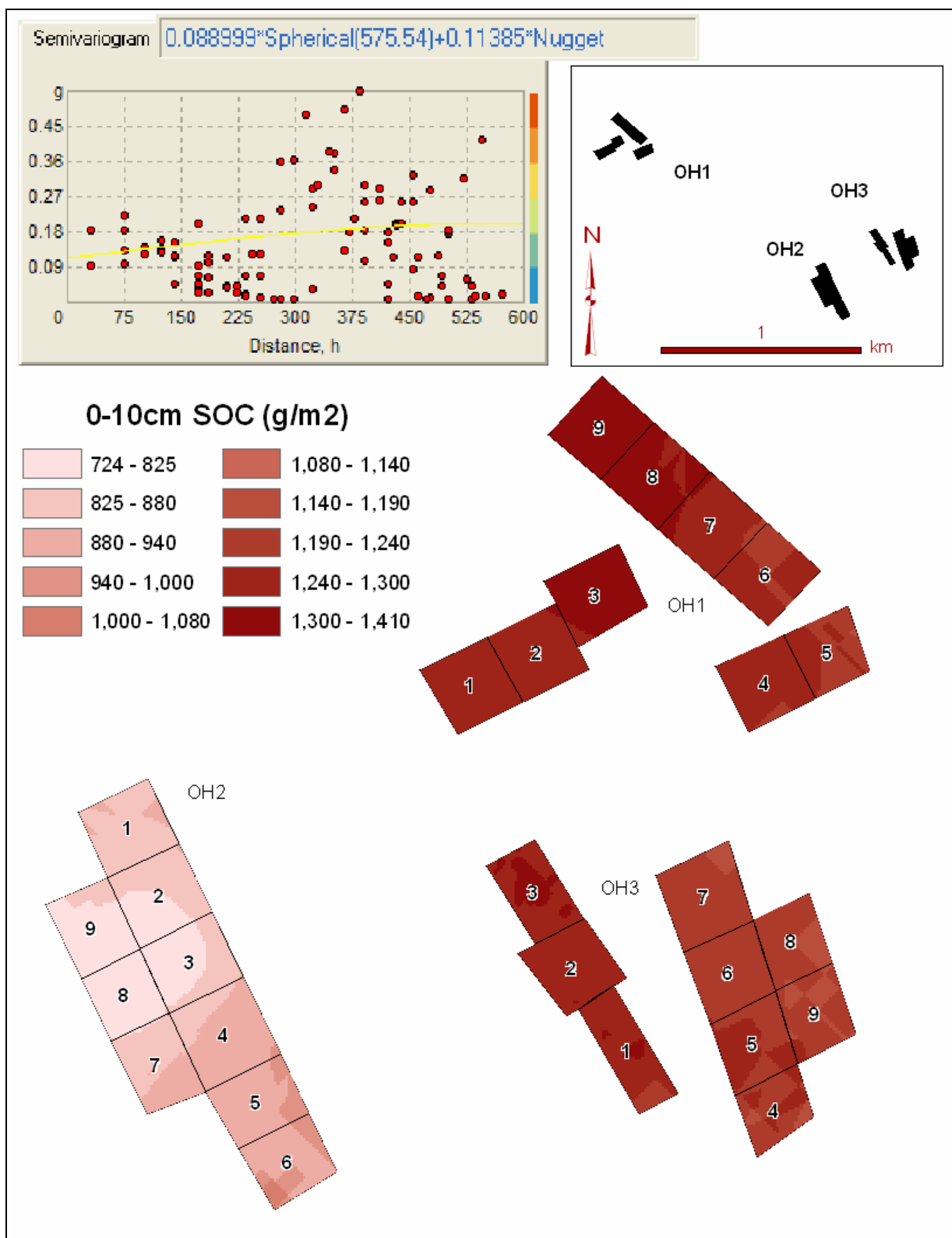


Figure 1-2-12. Horizontal distribution of SOC(g m⁻²) to the 0-10 cm soil depth on study sites in OH. The semivariogram function indicated presence of spatial dependence of SOC content from sampling locations up to 576 m apart across the mined landscape.

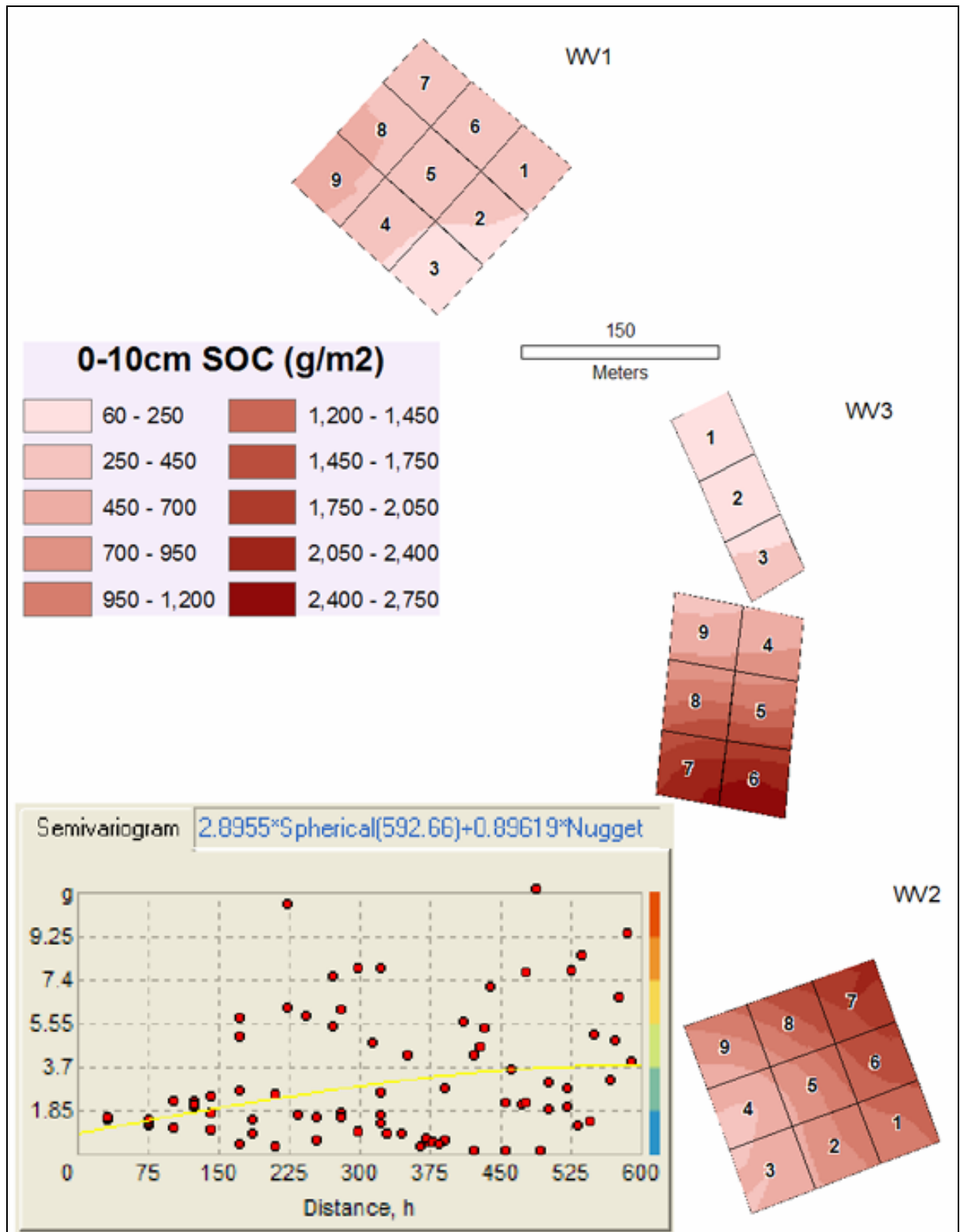


Figure 1-2-13. Horizontal distribution of SOC(g m⁻²) to 0-10 cm soil depth on study sites in WV. The semivariogram function indicated the presence of spatial dependence of SOC content from sampling locations up to 593 m apart across the mined landscape.

One of the benefits from using SOC(g m^{-2}) prediction maps developed by the ordinary kriging procedure was the ability to locate areas of low and high soil carbon sequestration rate. Our results indicated that the northernmost plots in the OH1 block in OH (plot numbers 3, 8, and 9) sequestered the highest carbon amount in the 0-10 cm depth ranging from 1,300 to 1,410 g C m^{-2} , and four of the central plots in OH2 (plot numbers 2, 3, 8, and 9) contained the lowest carbon amount ranging from 724 to 825 g C m^{-2} (Figure 1-2-12). Similarly, the southernmost plots of the WV3 block (plot numbers 6 and 7) in WV contained the highest amounts of sequestered SOC(g m^{-2}) on this site ranging from 2,050 to 2,750 g m^{-2} , and the northernmost three plots of the same block contained the lowest SOC(g m^{-2}) ranging from 60 to 250 g m^{-2} (Figure 1-2-13). Despite their spatial proximity of less than 300 m apart, the study plots in the WV3 block had completely different soil properties, which resulted in contrasting amounts of sequestered SOC so that the minimum carbon content was nearly one order of magnitude smaller than the maximum.

If the study sites in OH (or WV) were a part of a large-scale carbon sequestration project, the SOC prediction maps could be used by the landowner to apply selected management practices on the areas of different carbon sequestration potential so that resources could be utilized efficiently. The cumulative benefits from a targeted land management approach on mined sites, for which one of the goals is maximizing terrestrial carbon sequestration would exceed the costs of preliminary soil sampling, analysis, and mapping of the surface SOC content. Furthermore, if SOC maps were overlaid with available coarse-resolution FSQC maps, then the combined SOC and FSQC data would be sufficient to improve the accuracy, and increase the resolution of the original FSQC maps. The combined SOC and FSQC data could also be used to establish a correlation between forest site quality class and carbon sequestration rate for specific forest management treatments on mined land.

Spatial analysis of SOC data on mined lands could also be used to estimate the optimal sampling intensity for carbon inventories. For example, the semivariogram depicted in Figure 1-2-13 indicated that at a separation distance of 150 m between two sampling locations in WV, which is equivalent to sampling intensity of approximately 4 plots per 10 ha, the average standard deviation of SOC predictions (from the kriging map) would be approximately 1.38 g C m^{-2} (estimated as the square root of the semivariance at distance 150 m (estimated as the $\text{SQRT}(1.900)$) (Figure 1-2-13). Similarly, at a separation distance of 75 m, equal to a sampling intensity of approximately 18 plots per 10 ha, the average standard deviation of the predicted SOC values would be 1.16 ($= \text{SQRT}(1.344)$). These results indicated that for a 350% increase in sampling intensity (from 4 to 18 plots per 10 ha), there would be a much lower gain in accuracy of the SOC measurement on the sites in WV, approximately a 16% decrease in the standard deviation of predicted SOC.

The above example is similar to the argument made by Dollhopf (2000) who discussed the trade-off between increasing sampling intensity and the relative gain in precision of predicted soil properties on mined lands. Dollhopf (2000) concluded that geostatistical analyses for mined lands should become an important part of the planning for any reclamation task. Similar to the analyses in the above example, we recommend that geostatistical analyses also be used in the planning for carbon inventories on mined land carbon projects, for which accurate and precise SOC estimates are required in order to report and claim carbon credits.

Until geostatistical analysis methods become the new standard for SOC analysis on mined

land, standard statistical methods could be used to determine the optimal sampling intensity for SOC estimation. The sampling intensity for a certain project area could be determined using the mean and the standard deviation of SOC content on mined land as follows (Crepin and Johnson, 1993):

$$N = \frac{t^2 s^2}{D^2} \quad \text{Eq.2}$$

where t = number from the *Student's t* table (SAS, 2007) for a chosen level of precision (e.g. 95% probability level); s^2 = variance of the data, which must be known beforehand (from preliminary data sampling), or could be estimated as $(R/4)^2$, where R is the estimated range of SOC content likely to be encountered during sampling (Freese 1962); D = the variability that one would be willing to accept in mean estimation (e.g. within 10% of the mean). The relationship between sampling intensity and SOC variation (i.e. higher variance) depicted in Eq.2 indicates that proportionally larger number of soil samples would be required to achieve a certain accuracy of SOC estimates measured on sites with higher horizontal variation compared to sites with lower variation.

Consider the variation of the three mined sites in OH (Table 1-2-2). These results indicate that the SOC(g m⁻²) content measured on the OH2 research block was significantly lower than the SOC estimates for the other two research blocks (OH1 and OH3). The OH2 block was also characterized with the highest CV (44%) compared to OH1 (27%) and OH3 (30%). Using the relationship in Eq. 2, we estimated that 422% more samples would have to be excavated and analyzed on the OH2 block (35 plots ha⁻¹) compared to the OH1 and OH3 blocks (6.7 plots ha⁻¹), in order to measure SOC(g m⁻²) for the 0-10 cm depth with accuracy level of 10% within the true mean at the 95% probability level (precision) (Fig. 1-2-14). Similarly, 280 and 135 plots ha⁻¹ would be the respective sampling intensities for SOC analysis on each of the research blocks in VA and WV necessary to achieve 10% accuracy of the mean at the 95% probability level. Evidently, sampling of such high intensity is impractical and undesired for any carbon sequestration analysis, especially for rocky and compacted mine soils.

Therefore, most researchers would prefer lower accuracy and precision levels of reported SOC estimates in order to minimize the associated analytical costs. For example, 57% reduction of the sampling intensity in OH2 would result in 15% reduction in the precision of the estimated SOC(g m⁻²), from 95% to 80%. Similarly, if the probability level were kept at 95%, the same reduction in sampling intensity would lead to a 5% reduction in the accuracy of the estimated SOC(g m⁻²), from 10 to 15% within the mean (Fig. 1-2-14). However, reduction in the accuracy or precision of SOC estimates may deem these estimates unacceptable to claim for carbon credits, since these results would have wider confidence limits compared to SOC estimates of higher accuracy and precision levels.

But how low would a sampling intensity be? To provide an answer to this question, we determined the MCD, MDD, and the minimum number of years before carbon inventories should be scheduled in the future, so that, at acceptable accuracy and precision levels, the reported SOC(g m⁻²) estimates would indicate statistically significant SOC accumulation.

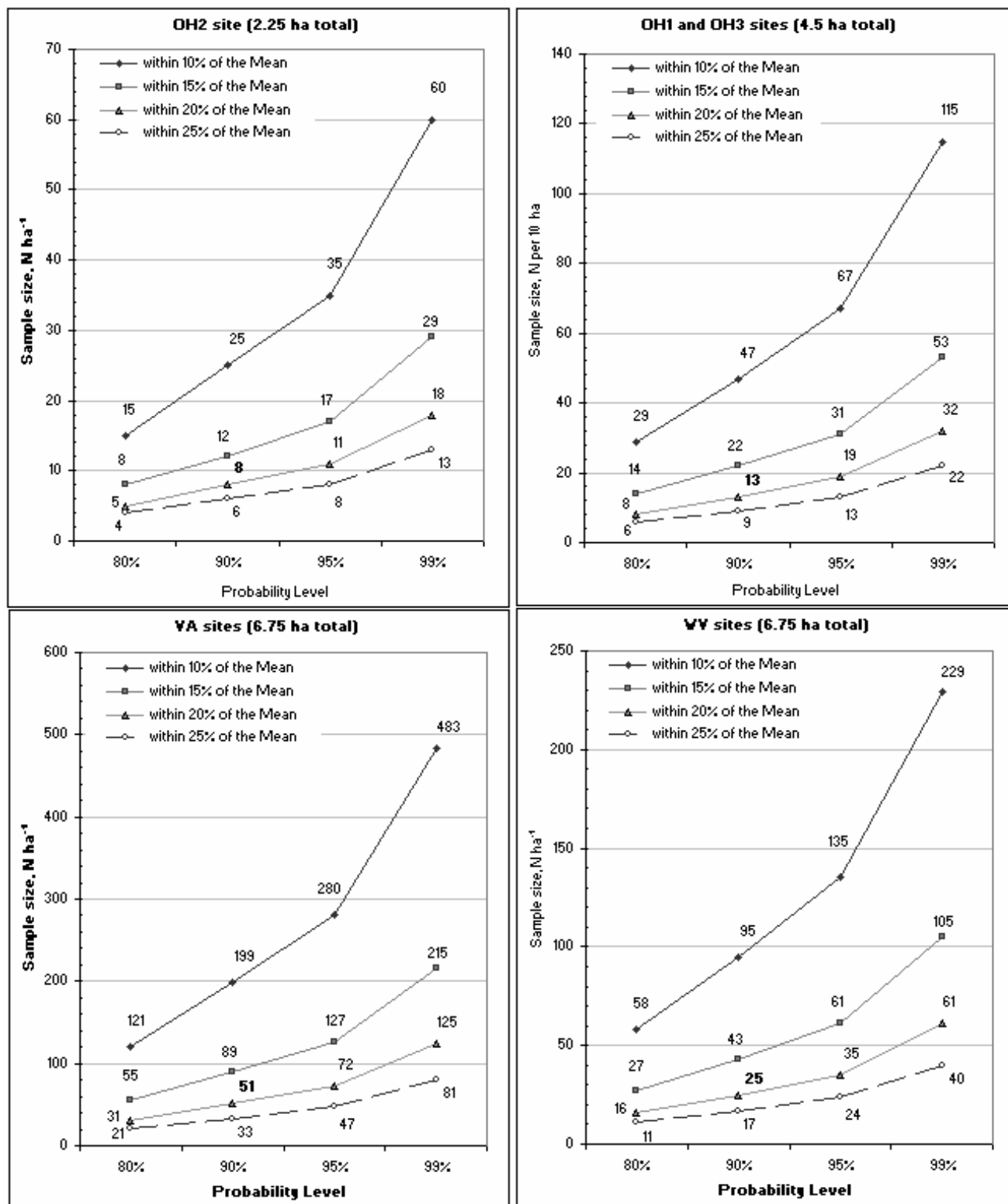


Figure 1-2-14. Sampling intensity estimates for SOC(g m⁻²) determination on mined lands in OH, WV, and VA computed at four probability levels (80, 90, 95, and 99%) and four accuracy levels (10, 15, 20, and 25% from the mean) of analysis.

MDD of Carbon Stocks and MCD for Carbon Inventorying on Mined Land

The concept of MDD of carbon stocks measured at two different points in time for a given project site has been the focus of many carbon sequestration studies (Conant et al. 2003; Lilienfein et al. 2003; Zhang and McGrath 2004). Carbon researchers are aware that in order to claim carbon credits, the total sequestered SOC(g m^{-2}) should be significantly (statistically defined) higher than the SOC(g m^{-2}) estimated at a previous carbon inventory event from 5-10 (or more) years ago. In some cases, authors reported statistically insignificant accumulations of SOC even after 30 years under a grassland land-use system (Zhang and McGrath, 2004). Others have reported statistically significant accumulations after 4 years for cultivated lands and forested systems (Conant et al., 2003).

It is evident that as soil variability increases within a certain terrestrial system, the ability to detect significant changes in SOC using a set number of samples decreases. This relationship was demonstrated by Conant et al. (2003) who showed that a greater number of sampling plots are required in forested systems compared to cultivated areas in order to detect a 2 Mg C ha^{-1} carbon stock change over 4 years. However, an increase in the number of sampling plots for highly variable soils such as mine soils could lead to a dramatic increase in sampling costs for carbon verification studies. Hence, the MCD of soil sampling on mined land could decrease (become shallower) proportionally due to the fact that soil excavation for SOC analysis is a rather tedious and expensive task, especially in compacted and rocky mine soils.

A more useful graphical representation of the vertical distribution of SOC(g m^{-2}) on the sites in OH (presented in Figure 1-2-7b) is depicted in Figure 1-2-15b. The cumulative SOC content is presented as percent of the total SOC contained within the 0-200 cm soil pedon (Figure 1-2-15b); we used the data in this graph to determine the MCD for SOC inventorying for the study sites in OH, and to present the computational steps for MCD estimation for other mined lands. The results from the mined sites in OH indicated that about 33% of the total-profile SOC was found in the surface 0-47 cm soil layer, while more than 66% was located in the 0-120 cm layer. Despite the fact that soil excavation on these sites to a greater soil depth would indicate that greater amount of carbon is sequestered in the soil, in compacted and rocky mine soils, there is a specific soil depth beyond which soil sampling for SOC analysis becomes cost-prohibitive.

Due to the increase of the amounts of excavated soil mass necessary for soil sampling to a greater soil depth, it is clear that the carbon credits from “deeper” SOC would be of lesser net value (difference between carbon credit profits and soil sampling/analysis costs) due to the higher costs of soil sample collection. We used the vertical distribution and variability of SOC down the soil profile, and the variability of SOC across the mined landscape to estimate MCD.

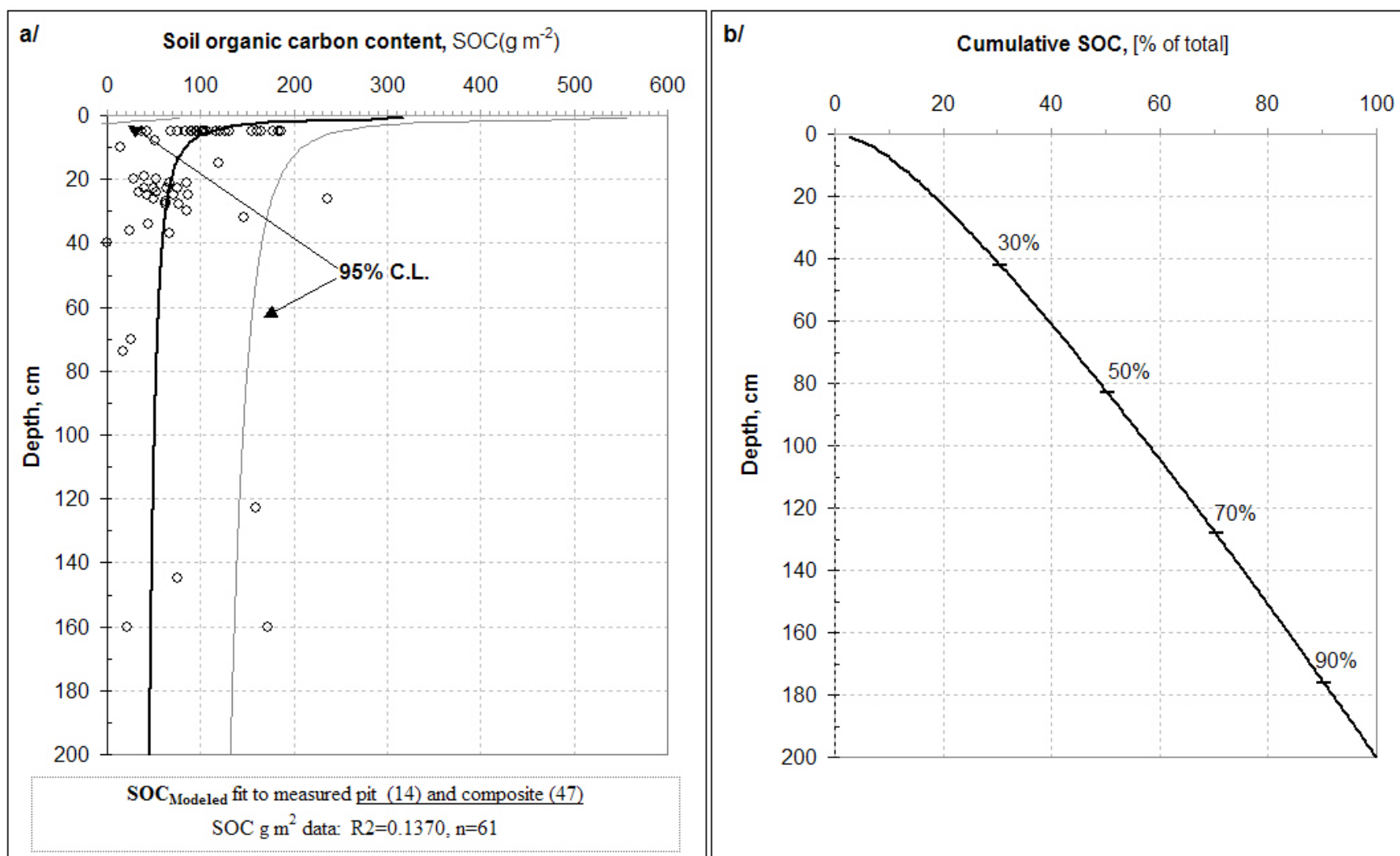


Figure 1-2-15. Vertical distribution of (a) SOC(g m⁻²) (estimated from data in Figures 1-2-6 and 1-2-7a using Eq. 1), and (b) cumulative distribution of SOC expressed as percent of the total SOC(g m⁻²) sequestered down the profile within the 0-200 cm soil pedon of eight deep pits excavated in OH. Thicker line indicates the fit of a statistically significant prediction model to the data, which are marked with circles, and lighter lines show the 95% confidence limits of these predictions.

Assume the following cost scenario: (i) at a sampling intensity of 3 soil pits ha^{-1} , the total costs for soil excavation of a 5-cm-thick soil layer within a mine soil profile, sample preparation, and carbon analysis were \$40 per soil pit. Assuming that sampling/digging costs of 3 pits were \$15, sample preparation costs for three samples in each 5-cm layer in each of the 3 pits were \$63, and SOC analysis costs for the nine samples were \$5 each, a total of \$120 would be the cost for SOC estimation in a 5-cm layer in 3 pits per ha. Similarly, assume total profits for sequestering atmospheric CO_2 in the soil at $\$127.2 (\text{Mg C})^{-1}$, based on the market price of carbon reported at the European Carbon Market (<http://www.pointcarbon.com>) on April 9, 2005. For the above cost/profit scenario, the total costs for measuring the sequestered $\text{SOC}(\text{g m}^{-2})$ within a 1-cm soil layer across 1 ha of project area on mined lands would be $\$24 \text{ cm}^{-1} \text{ ha}^{-1}$ (equal to $\frac{\$40}{5\text{cm} * 1\text{pit}} * \frac{3\text{pits}}{1\text{ha}}$); or $\$8 \text{ cm}^{-1} \text{ pit}^{-1}$ at an assumed sampling intensity of 3 soil pits ha^{-1} .

Now consider the sampling intensity results for the OH2 study site in OH, for which 17 soil pits ha^{-1} are necessary to produce SOC estimates at 15% accuracy and 95% probability levels (Figure 1-2-14). The total costs for SOC analysis on the OH2 site at this sampling intensity would be $\$136 \text{ cm}^{-1} \text{ ha}^{-1}$ (equal to $\frac{\$40}{5\text{cm} * 1\text{pit}} * \frac{17\text{pits}}{1\text{ha}}$). Therefore, in order for a SOC inventory to be feasible below a certain soil depth, the costs for SOC measurement on this site must be lower than the C-credit profits from the measured $\text{SOC}(\text{g m}^{-2})$; i.e., the profit-to-cost ratio $\$(\text{SOC}_{\text{Credit}})/\$(\text{SOC}_{\text{Inventory}})$ must be greater than 1 (Table 1-2-3).

For selected soil depths down the profile, the results in Table 1-2-3 depict the computational steps leading to the identification of the MCD for SOC analysis on the OH2 study site in OH; these steps could be used for MCD estimation on any mined land provided the vertical SOC distribution and the horizontal sampling intensity are known from preliminary soil analyses. In column *b*, we presented the $\text{SOC}(\text{g m}^{-2})$ values for a 1-cm-thick layer at selected depths (in column *a*) within the soil profile, while in columns *c* and *d*, respectively, we presented the cumulative SOC content as g C m^{-2} and as percent of the total-profile $\text{SOC}(\text{g m}^{-2})$.

For example, the second data row shows that the SOC for the 9-10 cm soil layer was 83.85 g C m^{-2} , and that the cumulative SOC for the 0-10 layer was $1.339.08 \text{ g C m}^{-2}$, which represented 11.8% of the total-profile SOC content. Note that a graphical representation of the data in columns *b* and *d* are also depicted in Figures 1-2-15a and 1-2-15b, respectively.

Because of the logistics of soil excavation, before collecting soil samples from soil depths greater than 50 cm, the soil mass from the 0-50 cm layer must be excavated first, which is also associated with certain excavation costs. We created a new SOC variable in column *e* (Table 1-2-3) to represent the amount of $\text{SOC}(\text{g m}^{-2})$ standardized by excavation depth, (column *e*) = (column *c*) / (column *a*). The profit-to-cost ratio depicted in column *f* (Table 1-2-3) was estimated as (column *e*, $\text{g C m}^{-2} \text{ cm}^{-1}$) $\times (1/100) \times (\$127.2 \text{ Mg}^{-1} \text{ C}) / (\$136 \text{ cm}^{-1} \text{ ha}^{-1})$; the division by 100 was used to convert g C m^{-2} units to Mg C ha^{-1} , which was necessary for this computational step. Using the profit-to-cost ratio estimates in Table 1-2-3, we determined the soil depth to which SOC inventories would be economically feasible, so that, at the very least, the carbon credits received for sequestering atmospheric CO_2 in the mine soil would be enough to recover the costs for measuring and reporting carbon credits (i.e., the ratio should be greater or equal to 1) (Table 1-2-3).

Table 1-2-3. Identification of the MCD for SOC(g m^{-2}) inventory on the OH2 study site in OH at carbon market price equal to $\$127.2 (\text{Mg C})^{-1}$, and total costs for SOC analysis at 15% accuracy and 95% probability levels equal to $\$8 \text{ cm}^{-1} \text{ pit}^{-1}$ at sampling intensity estimated at 17 pits ha^{-1} .

Depth	SOC	Cumulative SOC	SOC standardized by excavation depth		Profit-to-Cost ratio
a	b	c	d	e	f
[cm]	[g m^{-2}]	[g m^{-2}]	[%]	[$\text{g m}^{-2} \text{ cm}^{-1}$]	---
1	316.43	316.43	2.8	316.43	2.96
10	83.85	1,339.08	11.8	133.91	1.25
17	71.80	1,871.43	16.5	110.08	1.03
18	70.76	1,942.19	17.1	107.90	1.01
19	69.81	2,012.00	17.7	105.89	0.99
20	68.94	2,080.94	18.3	104.05	0.97
30	62.98	2,734.39	24.1	91.15	0.85
40	59.48	3,343.69	29.4	83.59	0.78
50	57.07	3,924.63	34.5	78.49	0.73
60	55.26	4,485.03	39.5	74.75	0.70
70	53.82	5,029.47	44.3	71.85	0.67
80	52.63	5,560.95	48.9	69.51	0.65
90	51.61	6,081.53	53.5	67.57	0.63
100	50.73	6,592.74	58.0	65.93	0.62
110	49.96	7,095.74	62.5	64.51	0.60
120	49.27	7,591.45	66.8	63.26	0.59
130	48.64	8,080.61	71.1	62.16	0.58
140	48.07	8,563.82	75.4	61.17	0.57
150	47.55	9,041.59	79.6	60.28	0.56
160	47.06	9,514.35	83.7	59.46	0.56
170	46.61	9,982.47	87.9	58.72	0.55
180	46.19	10,446.27	91.9	58.03	0.54
190	45.80	10,906.01	96.0	57.40	0.54
200	45.43	11,361.96	100.0	56.81	0.53

The results indicated that the profit break-even depth (i.e., MCD) of SOC analysis for carbon credits on the OH2 research block was 18 cm at a sampling intensity of 17 plots ha^{-1} , which allowed SOC estimation at the 15% accuracy and 95% probability levels (Figure 1-2-16a, Table 1-2-3). That is, despite the fact that 80% of the total amount of sequestered SOC was located below 18 cm depth on this site (Figure 1-2-15b, Table 1-2-3), one would be losing money if SOC was measured beyond this depth. In comparison, due to the lower horizontal variability of the OH1 and OH3 sites, the MCD for SOC estimation at the 15% accuracy and 95% probability levels on these two sites exceeded 200 cm (Figure 1-2-16b).

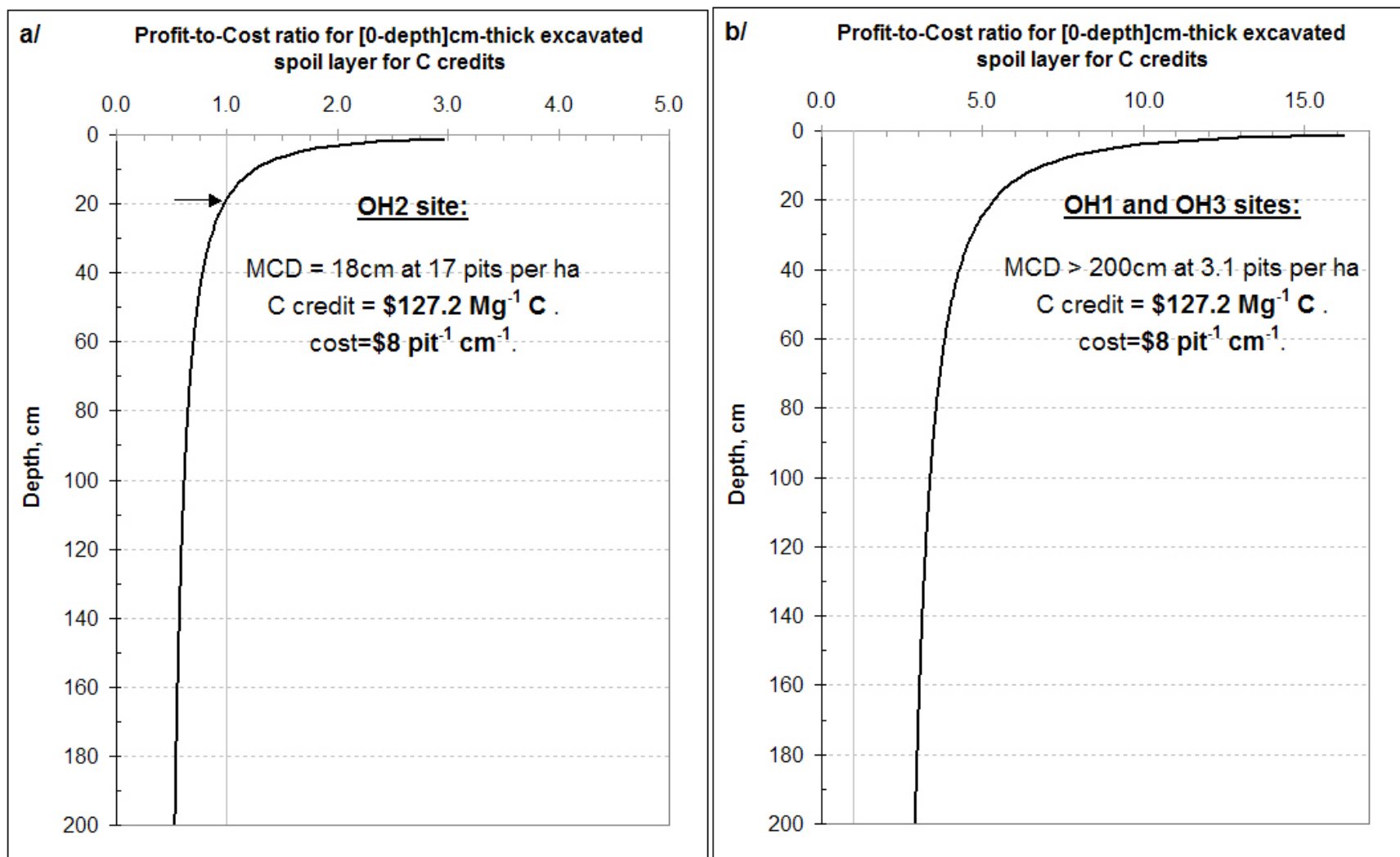


Figure 1-2-16. Maximum cost-effective depth and profit-to-cost ratios for SOC inventory on the (a) OH2 and (b) OH1 and OH3 sites in OH estimated at a carbon market price equal to \$127.2 (Mg C)⁻¹, total costs for SOC analysis at \$8 cm⁻¹ pit⁻¹, an sampling intensities at 17 and 3.1 pits ha⁻¹ for OH2, and for OH1 and OH3 sites, respectively.

We used the data depicted in Figure 1-2-16 to estimate the net gain from carbon credits, as well as the net costs for SOC analysis. For example, if SOC was analyzed to 0-25 cm depth on the OH1 and OH3 sites, 21% of the total-profile SOC(g m^{-2}) would be reported for carbon credits, which was equivalent to $2,386 \text{ g C m}^{-2}$ (equal to $0.21 \times 11,362 \text{ g m}^{-2}$) (Figure 1-2-16b); reporting the latter carbon amount would result in a gross profit of $\$3,035 \text{ ha}^{-1}$, based on the market price of carbon at $\$127.2 (\text{Mg C})^{-1}$. The total costs for measuring and reporting the SOC(g m^{-2}) to 0-25 cm depth on these sites would be $\$620 \text{ ha}^{-1}$ (equal to $\$8 \times 3.1 \times 25 \text{ cm}$) at sampling intensity of 3.1 pits ha^{-1} and assumed costs for SOC analysis at $\$8 \text{ cm}^{-1} \text{ pit}^{-1}$ (Figure 1-2-16b). By subtraction, the net profit from carbon credits to 0-25 cm depth on the OH1 and OH3 sites would be approximately $\$2,400 \text{ ha}^{-1}$. Similarly, a SOC analysis to 0-100 cm depth would yield net profit of about $\$5,900 \text{ ha}^{-1}$, estimated as the difference between the gross profits at $\$8,380 \text{ ha}^{-1}$ and gross costs at $\$2,480 \text{ ha}^{-1}$, which would be spent before receiving the profits from carbon credits.

It should be noted that the theoretical profit-to-cost analysis for the study sites in OH, described above, would be valid if the measured carbon content was sequestered within the timeframe of a carbon sequestration project established on these sites. However, in this case, the reported amounts of sequestered SOC(g m^{-2}) by soil depth in Table 1-2-3 are the baseline carbon estimate for these sites, which would be subtracted from the SOC(g m^{-2}) values measured in the future.

The computational steps used to determine the MCD in Table 1-2-3 are intended for use by mined land owners to aid in their choice of an economically-feasible sampling scenario that would best suit their goals. These computational steps could also be used to compute an approximation of the net profits that would be gained if the mined land was managed for carbon sequestration. It is important that all analyses to determine the MCD and profit-to-cost ratio for SOC measurement be performed after the baseline SOC content is subtracted from the SOC data for a future carbon inventory event.

Another useful piece of information for mined land owners would be the minimum number of years before a future carbon inventory event is feasible so that the difference [$\text{SOC}(\text{g m}^{-2})_{\text{future}} - \text{SOC}(\text{g m}^{-2})_{\text{baseline}}$] is statistically significant at the 95% probability level. Although SOC is expected to accumulate yearly during the life of a carbon sequestration project, due to the variability of SOC in mine soils, and the random errors associated with measuring SOC in soil samples, the detectable difference [$\text{SOC}(\text{g m}^{-2})_{\text{future}} - \text{SOC}(\text{g m}^{-2})_{\text{baseline}}$] would become statistically significant after a certain number of years of uninterrupted SOC accumulation; this difference is what we refer to as the MDD. In the following discussion, the number of years necessary to detect significant changes in SOC accumulation on a certain mined site is referred to as “significant” years until future carbon inventory (Figure 1-2-17).

We estimated the upper 95% confidence limit of the cumulative SOC(g m^{-2}) content for the mined sites in OH to illustrate the increase in variability of SOC estimates at 1-cm increments down to 200 cm soil depth (Figure 1-2-17a). The results clearly show that the variability of the SOC estimates at shallower depths is much smaller than that for deeper layers (Figure 1-2-17a). In order to make a decision as to when to schedule the next carbon inventory event, one should also know the expected variability (lower 95% confidence limit) of the future SOC content down to 200 cm, or at least down to the estimated MCD for the site.

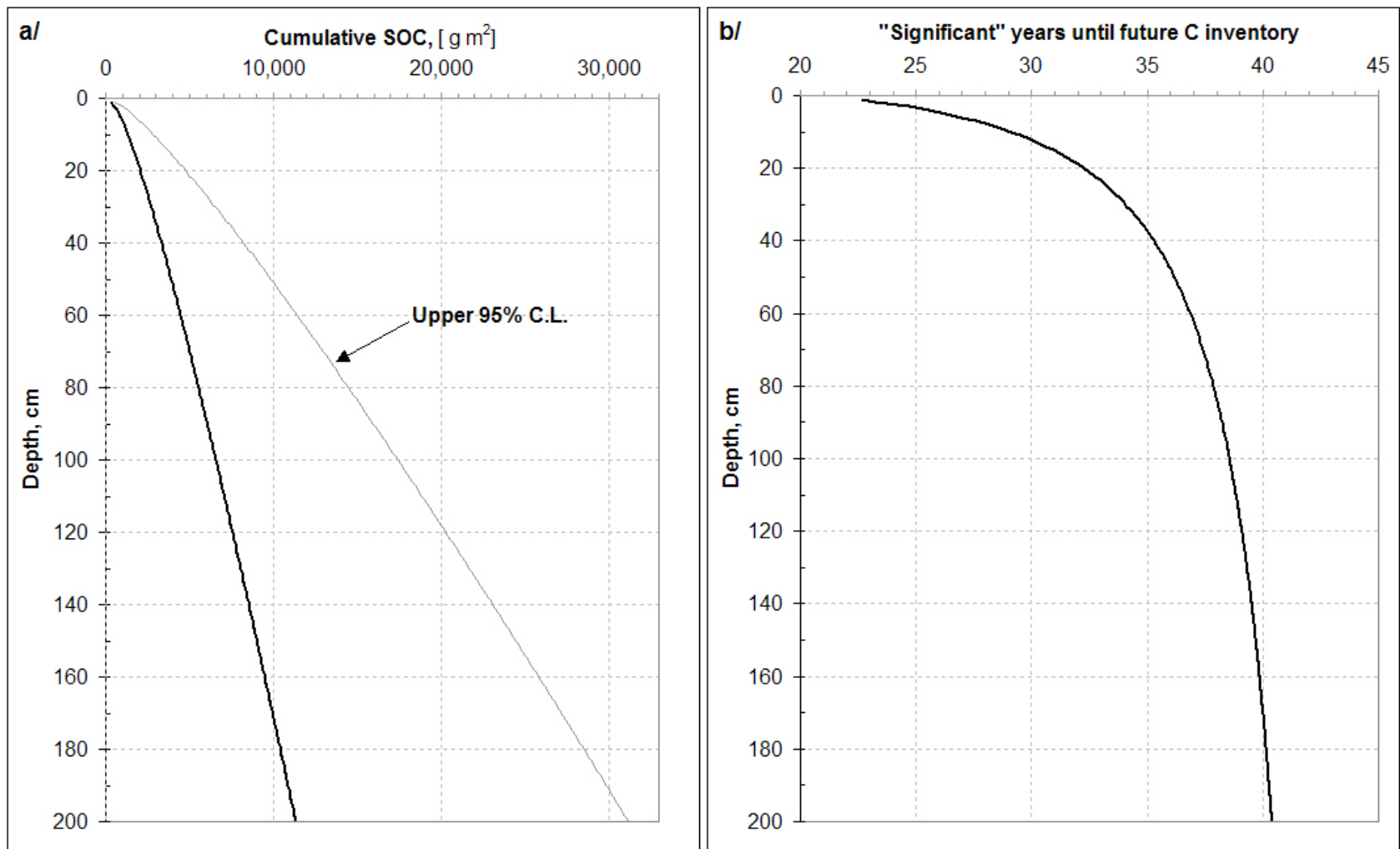


Figure 1-2-17. Minimum number of years required for the cumulative SOC(g m⁻²) content from two consecutive carbon inventories to a certain depth on mined sites in OH to be significantly (in statistical terms) different from each other at the 95% probability level.

In reality, it is impossible to know exactly what the SOC variability would be, but a good approximation for this estimate could be made based on the following assumptions: (i) the rate of SOC accumulation is a constant ($\text{g C m}^{-2} \text{ yr}^{-1}$) (Heath et al. 2002), and it can be estimated by dividing the present SOC content by the age of the site, and it would stay the same during a period of T years between the present and a future carbon inventory event; and (ii) the upper 95% confidence interval (C.I.) (computed as the difference between the value at the upper 95% confidence limit and the value at the mean) of the present SOC(g m^{-2}) content is equal to the lower 95% C.I. of the future SOC(g m^{-2}) content. That is, we assumed equal estimates of the variabilities around both the present and future SOC(g m^{-2}) contents measured on a certain mined site at two consecutive carbon inventories.

We used the mean cumulative SOC(g m^{-2}) estimate to determine the rate of SOC accumulation within a soil layer of certain thickness ([0-depth] cm, where depth ranged from 0 to 200 cm). For example, the average rate of SOC accumulation within the 0-20 cm soil layer for the sites in OH was estimated at $231 \text{ g C m}^{-2} \text{ yr}^{-1}$ as follows: $[2,081 \text{ g C m}^{-2} / 9 \text{ yrs}]$, where $2,081 \text{ g C m}^{-2}$ was the cumulative SOC content within the 0-20 cm soil layer, and 9 yrs was the approximate age of these sites (Table 1-2-3). Similarly, the rate of SOC accumulation within the total soil profile (0-200 cm) was estimated at $1,262 \text{ g C m}^{-2} \text{ yr}^{-1}$ ($= 11,362 \text{ g C m}^{-2} / 9 \text{ yrs}$).

A measure of the 95% C.I. for the study sites in OH was obtained from the data presented in Figure 1-2-17a. We estimated the 95% C.I. as the difference (95% C.L. estimate – Mean estimate) for the cumulative SOC(g m^{-2}) content to 200 cm depth. For example, the 95% C.I. of the cumulative SOC content within the 0-20 cm layer was estimated at $2,688 \text{ g C m}^{-2}$ as follows: $[4,769 \text{ g C m}^{-2} - 2,081 \text{ g C m}^{-2}]$, where $4,769 \text{ g C m}^{-2}$ was the upper 95% C.L. of the mean SOC content, which was $2,081 \text{ g C m}^{-2}$ (Figure 1-2-17a).

Based on the above two assumptions, we predicted the number of “significant” years until future carbon inventory as follows: $T = [\text{SOC}(\text{g m}^{-2})_{\text{at present}} + 2 \cdot (95\% \text{ C.I.})] / \text{rate}_{\text{at present}}$. Estimates for T to 200 cm soil depth are presented in Figure 1-2-17b. These results showed that approximately 32 years of uninterrupted SOC accumulation within the 0-18 cm soil layer on the OH2 mined sites would be necessary so that the future SOC estimate ($\text{SOC}_{\text{at [present + 32 yrs]}}$) is significantly larger than the SOC measured at the present carbon inventory; 18cm was the estimated MCD for the OH2 site (Figure 1-2-17b). The MDD of SOC(g m^{-2}) stock change in the 0-18 cm soil layer was estimated at $5,376 \text{ g C m}^{-2}$ (equal to $2 \cdot 95\% \text{ C.I.}$) after 32 years of uninterrupted SOC accumulation on this site.

The data presented in Figure 1-2-17b showed that any future SOC(g m^{-2}) inventory on the OH2 site that would be carried out to 18 cm depth sooner than 32 years into the future would yield SOC(g m^{-2}) estimates that would appear to be similar to the SOC(g m^{-2}) content measured at a present carbon inventory event; i.e., the SOC difference would be smaller than the MDD for this site, which would falsely indicate that there was no carbon sequestration in the soil. Unless the future carbon inventory is done at least 32 years into the future, there would be insufficient statistical evidence to prove that the SOC content to 18 cm soil depth on the OH2 site had increased compared to the SOC content measured at a present carbon inventory.

Clearly there is a need for more efficient carbon analysis techniques and more advanced methods and tools for horizontal soil variability analyses, such as geographic information systems and geostatistics, in order to minimize the costs for SOC analysis across the landscape and to focus the profits from carbon credits to expand current carbon sequestration projects.

Furthermore, the future involvement of the United States and other nations of the world in a global carbon market, similar to the market currently established in Europe, would increase the demand for carbon credits, which would lead to increase of the price of sequestered SOC. As a result, SOC analyses could become economically feasible to greater soil depths meaning that more thorough and complete SOC inventories may become possible. The greater good from the establishment of a global carbon market would be the increased interest in establishing ecosystems with high sequestration potential such as managed forests on disturbed land that would otherwise exist as abandoned grassland or fallow land.

It is an obligation of terrestrial ecosystem scientists to develop SOC measurement and analytical techniques to detect small accumulations of sequestered C, especially on mined lands of high landscape heterogeneity. Carbon inventory tools and techniques capable of detecting smaller MDD of SOC content on mined land would allow mined land owners to gain profits from carbon credits sooner than 30+ years, and use these profits to expand the land areas managed for carbon sequestration. Ultimately, these efforts would allow the expansion of terrestrial carbon sequestration projects and, thus, implementing one of the provisions of the Kyoto protocol, restoration of disturbed lands, to combat the negative effects of global warming (COP-UNFCCC 1997).

Baseline Ecosystem Carbon Content of Study Sites

The baseline carbon content for a certain mined site is estimated as the sum of sequestered carbon within the individual carbon pools of a certain ecosystem. For mined lands planted to grass, the carbon pools are the SOC, vegetation, and litter layer biomass.

Although there were sparsely-scattered shrubs (autumn olive) or small trees (black locust) across the mined sites in OH, VA, and WV, we did not estimate the amount of carbon contained within their vegetation biomass due to the lack of a valid and cost-effective method that could be used for rapid and accurate measurements of the biomass of shrubs and small-tree vegetation for carbon sequestration inventories. Compared to the biomass carbon contained within a dense stand of shrubs or forest trees of a similar land area, we assumed that the relative amount of sequestered carbon in the biomass of randomly scattered shrubs and small trees across mined sites was insignificant.

As mentioned earlier, data from the Flint Gap site was used to approximate the carbon stocks in the vegetation and litter layer carbon pools for the study sites in OH, VA, and WV. The land area of the Flint Gap site (about 35 ha) was nearly double the cumulative grassland area (about 20 ha) of all research blocks used in this study put together (Figure 1-2-1), representing a greater diversity of the composition and vigor for grass and legume species commonly sown on mined sites. The Flint Gap site was selected the project site for a large scale carbon sequestration project on surface coal-mined land, among many other reclaimed coal mine fields located in the Clinch and the Powell River valleys of southwestern VA, and was considered to represent most mined lands in the hard-rock region of the Appalachian coalfield. Based on the above information, we believe that the assumption for choosing the C_{biom} (Mg ha⁻¹) contents for the vegetation and litter layer carbon pools measured on the Flint Gap site to represent the baseline estimates of the above-ground carbon pools (vegetation biomass and litter layer) for the mined sites in this study was valid.

The flat areas of the Flint Gap site were seeded to a mixture of grass and legume species at the time of site reclamation, while the steeper slopes were planted to trees. The grass species that were observed on the site were annual ryegrass (*Lolium multiflorum* Lam.), orchard grass (*Dactylis glomerata* L.) and timothy (*Phleum pratense* L.), while the legumes were red clover (*Trifolium pratense* L.), white clover (*Trifolium repens* L.), *Sericea lespedeza*, and birdsfoot trefoil (*Lotus corniculatus* L.).

The vegetation and litter layer carbon pools on the Flint Gap site, which was 8 years old at the time of vegetation sampling during the summer 2005, were both assumed to be at the maximum potential for grassland mined sites (Table 1-2-4). This assumption was supported by data in the literature indicating that perennial grasses reach their maximum productivity potential, evident from a stable number of seedlings per unit area, within 4 years (or less) after seeding (Bakker et al. 2003; Bartholomew et al. 1995). Based on the above assumption, the respective above-ground carbon pools for all mined sites in OH (age 9) and WV (age 4) were estimated to be equal to the exact carbon contents in the vegetation and litter layer biomass measured on the Flint Gap. We estimated the carbon accumulation rates ($\text{g m}^{-2} \text{ yr}^{-1}$) in the vegetation and litter layer biomass for the Flint Gap site as $[\text{Mean\_C}_{\text{biom}}(\text{g m}^{-2})/4(\text{yrs})]$ (Table 1-2-4), which we used to determine the respective above-ground carbon pools for the mined sites in VA (age 1).

Table 1-2-4. Vegetation and litter layer $\text{C}_{\text{biom}}(\text{g m}^{-2})$ contents, and accumulation rates determined from a total of 429 samples collected from the Flint Gap mined site located in Russell County, VA. All samples were collected in summer 2005.

Type	Mean	Std.Error	Min	Max	N	Rate *
	----- g C m^{-2} -----					$\text{g C m}^{-2} \text{ yr}^{-1}$
Vegetation	87.65	3.347	16.36	386.06	218	21.91
Litter layer	80.29	3.729	2.53	279.99	211	20.07

* Indicates a constant carbon accumulation rate for 1- to 4-year-old mined sites; the C content for mined sites older than 4 years was assumed to be at the 4-year maximum, and to be equal to that of a 4-year-old site.

The baseline ecosystem carbon stock for the three sites in OH was estimated at $6,761 \text{ g C m}^{-2}$ (about $67.6 \text{ Mg C ha}^{-1}$) accounting for the $\text{SOC}(\text{g m}^{-2})$ content to 100 cm depth and the carbon content in vegetation and litter layer carbon pools. Because there were insufficient data for the vertical distribution of SOC for the study sites in VA and WV, we were not able to provide accurate measurements for the baseline ecosystem carbon content on these sites. However, assuming that the $\text{SOC}(\text{g m}^{-2})$ content of the analyzed surface soil layers (Table II-1) in VA and WV was the same for all layers down the soil profiles ranging from 10 to 100 cm depth for the respective sites, we used the results in Table II-1 to determine the best approximation of the $\text{SOC}(\text{g m}^{-2})$ stocks to 100 cm soil depth, and to estimate the baseline ecosystem carbon content.

For the sites in VA and WV, the baseline ecosystem carbon was estimated at 814 g C m^{-2} (equal to 8.1 Mg C ha^{-1}) and $3,253 \text{ g C m}^{-2}$ (equal to $32.5 \text{ Mg C ha}^{-1}$), respectively; these estimates included the $\text{SOC}(\text{g m}^{-2})$ to 100 cm depth and the $\text{C}_{\text{biom}}(\text{g m}^{-2})$ content in vegetation and litter layer biomass. For the sites in VA, we used an average $\text{SOC}(\text{g m}^{-2})$ for the 0-10 cm layer, which was weighted by the respective number of shallow pits that were reclaimed with and without a topsoil layer (Table 1-2-1).

Conclusions

Mine soil-specific sampling and analysis techniques such as the 16-step SOC estimation method, MDD, and MCD, are all necessary tools that could be used by mined land owners in order to warrant carbon sequestration projects on mined lands. An important outcome from this study was a three-tier data set including: (i) the soil carbon and vegetation biomass and litter layer carbon variability observed on mined sites sown to grass; (ii) MDD and MCD estimates for grass land mined lands; and (iii) the baseline carbon content for the study sites. These data sets can be incorporated in a decision support system for predicting carbon sequestration for forest establishment practices on mined lands.

Our findings indicated that in one growing season very little carbon was added to mine soils, evident from the observed low carbon amounts measured on the sites in VA ($\sim 10 \text{ g m}^{-2}$ within 1-cm layer) relative to the 9-year-old sites in OH ($\sim 87 \text{ g m}^{-2} \text{ cm}^{-1}$). The baseline ecosystem carbon stock, including soil carbon to 100 cm depth, for the three sites in OH was estimated at $6,761 \text{ g C m}^{-2}$, 814 g C m^{-2} and $3,253 \text{ g C m}^{-2}$ for the VA and WV sites, respectively.

We demonstrated that establishing carbon sequestration projects on mined lands could be of significant monetary benefit for the landowner, in addition to reducing atmospheric CO_2 levels. Assuming base costs of $\$8 \text{ cm}^{-1} \text{ pit}^{-1}$ for carbon analysis and base profits of $\$127.2 \text{ Mg}^{-1} \text{ C}$ from carbon credits, we determined that the net profit from measuring and reporting SOC contained in the 0-25 cm soil layer on the OH1 and OH3 sites was approximately $\$2,400 \text{ ha}^{-1}$, and the net costs were $\$620 \text{ ha}^{-1}$. Similarly, the net profits from SOC analysis for 0-100 cm layer were estimated approximately $\$5,900 \text{ ha}^{-1}$ at net costs of $\$2,480 \text{ ha}^{-1}$. We demonstrated that between 23 and 40 years of uninterrupted SOC accumulation within the mine soil profile on the OH2 mined sites are necessary for the future $\text{SOC}(\text{g m}^{-2})$ estimate to be significantly larger than the $\text{SOC}(\text{g m}^{-2})$ measured at the present time.

The overall findings from this study supported the understanding that establishing carbon sequestration projects on mined lands could be a profitable alternative for mined land reclamation. We presented a method to determine the minimum number of years before a future carbon inventory event is carried out so that the measured $\text{SOC}(\text{g m}^{-2})$ differences were statistically significant. We demonstrated that geostatistical analyses procedures could be used for analysis in mine soils, based on the estimated spatial-dependence predictability of surface $\text{SOC}(\text{g m}^{-2})$ data. We discussed the importance of using kriging techniques to create surface $\text{SOC}(\text{g m}^{-2})$ maps for the mined land managed for carbon sequestration. Due to the inherent variation of soil carbon in mine soils, we recommended that mined sites older than 9 years be selected for carbon sequestration projects.

Task 1, Sub-study 3: Protocol for Carbon Accounting and Monitoring

Experimental

To ensure consistency in the reporting and verification of carbon credits among the owners of forested mined lands, there is a need for comprehensive and practical guidelines for sampling, measurement, and analysis of carbon data for mined lands. Due to the anthropogenic origin of surface-mined lands, carbon accounting protocols for undisturbed lands are not suitable for mined lands. The objective of this study was to develop and evaluate a mined land-specific sampling protocol for ecosystem carbon analysis that would enable mined land owners to measure and report sequestered carbon in the form of vegetation and litter layer biomass, and soil organic carbon (SOC). We proposed a set of cost-effective and practical guidelines for ecosystem carbon accounting on mined lands located in the Appalachian coalfield using a four-step approach as follows: (step A) establishing permanent carbon monitoring plots across a mined site; (step B) baseline carbon stock estimation; (step C) periodic carbon content measurement; and (step D) reference areas. We established a prototype carbon sequestration project (about 75 ha area) located in Russell County, Virginia, to evaluate the proposed sampling and measurement protocol. The results from step A in the protocol are presented in this paper.

Detailed Field Sampling and Measurement Protocol for Carbon Inventory on Mined Land

Step A: Establishing permanent carbon monitoring plots.

A.1. Mined land site quality mapping:

At randomly selected locations across the entire extent of the project site, measure soil acidity (pH), EC, soil compaction, percent slope, aspect, soil texture, percent coarse rock fragment content, rock type, and maximum rooting depth using standard soil surveying techniques (Soil Survey Division Staff, 1993) and the mined-land specific methods described in Jones (2005). For each of these soil parameters, use the sufficiency curves reported in Jones (2005) to determine the general and the white pine-specific productivity indices using Eq. 1 and Eq. 2, respectively, and estimate the FSQC at each surveying location. Assign FSQC I to all strata with estimated PI greater-than-0.8; FSQC II to all strata with estimated PI ranging from 0.67 to 0.8; FSQC III to all strata with estimated PI ranging from 0.53 to 0.66; FSQC IV to all strata with estimated PI ranging from 0.39 to 0.52; and FSQC V to all strata with estimated PI less-than-0.38 (Jones 2005).

Using as a guide the type and vigor of the mined land vegetation observed in the field, and based on the FSQC estimates of each surveying location, delineate the boundaries of all mapping strata across the mined landscape so that surveying locations of similar FSQC are included in a larger, contiguous stratum. If the individual stratum's area extent is unreasonably small, such as < 0.04 ha (<0.1 ac), then include its area to a larger neighboring stratum of similar FSQC. Incorporating smaller mapping strata with larger ones could be based on the area-extent of the minimum land management unit, such as a minimum area of managed forest stands, which would be specific for each mined site.

A.2. Determining the number of soil sampling plots:

For each ecosystem carbon pool of a specific FSQC class on the project site, determine the sampling intensity using the mean and the standard deviation of carbon content estimates as follows (Crepin and Johnson, 1993):

$$N = \frac{t^2 s^2}{D^2} \quad \text{Eq.3}$$

where N = number of sampling locations per unit area, which is the area represented by all surveying locations of a similar FSQC; t = number from the *Student's t* table (SAS, 2007) for a chosen level of precision (e.g. 95% probability level); s^2 = variance of the data, which must be known beforehand (from preliminary data sampling), or could be estimated as $(R/4)^2$, where R is the estimated range of carbon content estimates likely to be encountered during sampling (Freese, 1962); D = the variability that could be accepted in mean estimation (e.g. within 10% of the mean).

In order to choose the most cost-effective sampling intensity for carbon content estimation within each FSQC class, N could be computed at three levels of accuracy (10, 15, and 20% within the mean) and at two levels of precision (90 and 95%). Due to existing uncertainties as to whether the plots would remain unchanged, undamaged, or intentionally manipulated between carbon inventories, select 20% more sampling plots per stratum (for each ecosystem carbon pool), which would be used as reserve sampling locations in addition to the sample size N.

Within randomly selected strata representing each identified FSQC class on the mined site, excavate shallow soil pits to 0-20 cm depths at 10 randomly-selected locations to determine the horizontal variation of soil organic carbon per unit area (SOC, g m^{-2}) (Fig. 1-3-1). It should be noted that 10 shallow pits are excavated for every individual FSQC class identified on the project site, as opposed to sampling a total of 10 sampling locations for the entire project site. The bulk density of fines, content of fines, and soil carbon concentration (C, wt%) are determined using the methods described in Substudy 2. The SOC(g m^{-2}) content to 0-20 cm depth for each soil pit is estimated using the following equation:

$$\text{SOC}(\text{g m}^{-2}) = C(\text{wt}\%) * \text{BD}_{\text{fines}}(\text{g cm}^{-3}) * \text{Fines}(\text{vol}\%) * \text{Layer}(\text{cm}) \quad \text{Eq. 4}$$

where $C(\text{wt}\%)$ = percent soil organic carbon of the fine soil fraction (<2 mm soil particles), measured on a weight basis, $\frac{C_g}{\text{Fines}_g} * 100$; $\text{BD}_{\text{fines}}(\text{g cm}^{-3})$ = bulk density of the fines measured as $\frac{\text{Fines}_g}{\text{Fines}_{\text{cm}^3}}$; $\text{Fines}(\text{vol}\%)$ = the volumetric fraction of the fines measured as percent of the total volume of the excavated soil sample, $\frac{\text{Fines}_{\text{cm}^3}}{\text{Soil}_{\text{cm}^3}} * 100$; $\text{Layer}(\text{cm})$ = soil layer or soil horizon thickness expressed in cm units.

The mean and standard deviation of SOC(g m^{-2}) from the shallow pits are used to determine the number of soil pits required to measure the amount of sequestered carbon on the mined sites for each stratum of a specific FSQC class. The soil-sampling intensities for each FSQC class are estimated using Eq. 3.

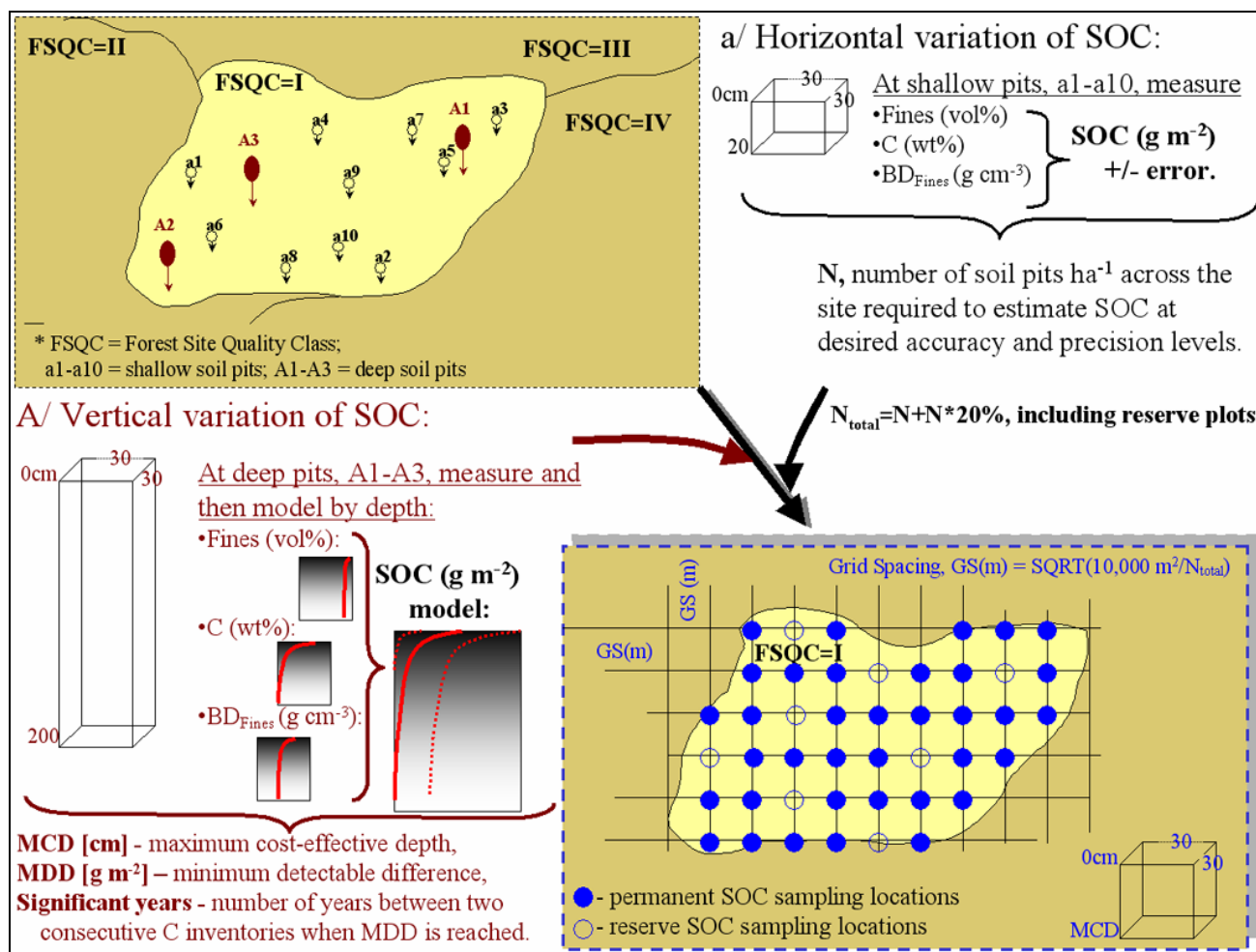


Figure 1-3-1. Soil organic carbon content sampling and analysis methods for a hypothetical mined site, which has been stratified to polygons of uniform FSQC.

Because soils could vary by depth, in addition to the shallow soil pits, excavate three to five deep pits down to 2 m depth in order to characterize the mine soil profile in the vertical dimension (Figure 1-3-1). Delineate the boundaries of each soil horizon using standard soil surveying methods (Soil Survey Division Staff, 1993) and collect bulk and loose soil samples for analysis. If soil horizons are not readily discernible, then the entire soil profile is sampled in 5-cm increments down to 200 cm depth. For each soil horizon, measure the bulk density of the fines, volumetric content of the fines, soil carbon concentration, and soil horizon depths using the methods described in Substudy 2.

Using the analytical approach described in Substudy 2 and the standard statistical tools (PROC REG, Cp model selection procedure in (SAS, 2007) in SAS® software, develop regression models to describe the distribution of BD_{fines} ($g\ cm^{-3}$), Fines(vol%), and C(wt%) by soil depth (Figure 1-3-1). Note that separate models of these three soil properties as a function of soil depth are required for each mapped FSQC class. Therefore, if the entire project area was mapped to multiple strata of two FSQC classes, then six to ten deep pits should be excavated at representative locations across the site.

The resulting models of SOC(g m⁻²) as a function of soil depth are estimated by multiplying the individual predictions for BD_{finer}(g cm⁻³), Fines(vol%), and C(wt%) in Eq. 4. The SOC(g m⁻²) models are used to determine the MCD, MDD, and the number of years in the future when the SOC accumulation in the soil would be significantly greater (in statistical terms) than the present SOC levels; the latter is also referred to as the number of “significant” years (Substudy 2) (Figure 1-3-1). The MCD, MDD, and “significant” years are estimated separately for each identified FSQC on the project site using the computational steps described in Substudy 2.

A.3. Determining the number of vegetation sampling plots:

Using an approach similar to the sampling intensity computation for the soil carbon pool, estimate the sampling intensities for the vegetation and litter layer carbon pools. Within randomly selected strata representing each identified FSQC class on the site, vegetation and litter layer biomass samples are collected at 10 randomly selected locations. The vegetation biomass at 5 cm (~2 in) above the ground surface is clipped from a square plot of 0.25 m² area. For sampling locations with more vigorous vegetation, a 0.10-m² clipping area is recommended in order to fit all clipped vegetation in the bag. The vegetation and litter layer samples are dried at 65°C, and their net weight is measured and recorded. Every 10th randomly selected sample bag, one for the vegetation and another for the litter layer samples, is ground to < 1 mm particle size for biomass carbon content analysis; an approximately 0.300-g subsample of each ground material is used for carbon content analysis. Using the dry sample weights, carbon content measurements, and the precise sample-clipping area, the amount of carbon sequestered in vegetation and litter layer per unit area (C_{biom}, g m⁻²) is computed as follows:

$$C_{biom}, g m^{-2} = (C_{biom}wt\%) * \left(\frac{Biomass_g}{0.25m^2} \right) * 100^{-1} \quad \text{Eq. 5}$$

where C_{biom}(g m⁻²) = carbon content per unit area contained within the vegetation or litter layer biomass; C_{biom}(wt%) = percent carbon contained within the dry vegetation or litter layer biomass, $\frac{C_g}{Biomass_g} * 100$; Biomass(g) = dry-weight of the vegetation and litter layer biomass; 0.25 = sampling plot area (m²) from which the biomass samples were collected; 100⁻¹ = unit correction factor.

The mean and standard deviation estimates for C_{biom}(g m⁻²) of the vegetation and litter layer biomass are used to determine the sampling intensities of the respective biomass types using Eq. 3.

A.4. Locating sampling plots on the field:

Overlay a square grid across the topography of the project site so that grid intersections become the centers of the sampling plots. For each ecosystem component within a stratum, the necessary sampling plots (N_{total}), including at least 20% more reserve plots, are located on the field at random (Figure 1-3-1). The spacing of the grid (in meters) is estimated as SQRT(10,000/N_{total}), where 10,000 m² = 1 ha, and SQRT is the square root function. Plot center locations are marked clearly with pin-flags and their location coordinates are recoded with a GPS unit. All plot locations are further secured with rebar, plastic pipes, and flagging tape by the field crew prior to collecting field data for the first carbon content inventorying event. These plots are termed permanent sampling plots for carbon measurement and monitoring in future

inventories.

Step B: Baseline carbon stock estimation.

B.1. Soils:

At each permanent soil-sampling plot, one soil pit is excavated with minimum horizontal dimensions at 30x30 cm and depth at the estimated MCD (Figure 1-3-1). Standard soil survey techniques are used to obtain horizon depths (Soil Survey Division Staff, 1993) from 0 to MCD. Loose samples are collected from each horizon for carbon analysis. If there are no clearly visible soil horizons, then soil samples are collected in 5-cm increments down to the estimated MCD.

Soil bulk density and percent coarse rock fragments are measured by the methods employed by Jones (2005). Loose soil samples are air-dried and passed through a 2-mm mesh sieve. Soil carbon concentration $C(\text{wt}\%)$ is measured using the 16-step SOC estimation method (Substudy 1). For each sampling location, the $\text{SOC}(\text{g m}^{-2})$ content is computed using Eq. 4. The average of all plot estimates within a stratum represents the baseline carbon content in the mine soil.

B.2. Vegetation:

The sampling and estimation methods used for the baseline carbon analysis are similar to those used for the preliminary sampling and estimation described in Step A (sub-step A.3.). At each permanent vegetation-sampling plot, vegetation biomass is collected from a 0.25 m^2 square area by clipping all live vegetation biomass at 5 cm from the ground and placing it in paper bags. For strata with more vigorous vegetation, a 0.10-m^2 clipping area is recommended in order to fit all clipped vegetation in the bag. For each sampling location, using the dry biomass weight, $C_{\text{biom}}(\text{wt}\%)$ estimates, and the precise sample-clipping area, the $C_{\text{biom}}(\text{g m}^{-2})$ content is estimated using Eq. 5. The average of all plot estimates within a stratum represents the baseline carbon content in the vegetation biomass.

B.3. Litter layer:

Use the sampling and estimation methods for the vegetation biomass carbon pool, described in step B (sub-step B.1.). The average of all plot estimates within a stratum represents the baseline carbon content in the litter layer biomass.

Step C: Periodic carbon content measurement.

Unless the sampling intensity is changed (reduced or increased) based on changes of the variation of carbon content estimates from the previous carbon inventory, sampling is done on all permanently marked sampling plots designated for a specific ecosystem component. Carbon inventories for the forest (overstory and understory canopy layers) and the grass vegetation are carried out every 5 years up until age 15 of the forest stand, and every 10 years thereafter. The soil carbon inventory is carried out no earlier than the number of “significant” years estimated in step A (sub-step A.2.).

C.1. Soils.

Because of the destructive nature of soil sampling, soil pits are excavated at different 30 x 30 cm horizontal locations for each carbon inventory event changing at a clockwise direction around the center of the permanent soil sampling plot so that the first soil pit (at 1st carbon inventory) is to the north of the plot center, the second soil pit (at 2nd carbon inventory) is to the

east, etc. Assuming that the time period between SOC inventories could exceed 30-40 years in order to detect significant carbon accumulations in mine soils (Substudy 2), depending on the heterogeneity of each project site, soil-sampling locations could be re-used after four consecutive inventories, or after approximately 120 years. The specific sampling and measurement tasks that are made at each permanent soil-sampling plot are as follows:

1. Mark the soil pit location for the current carbon inventory:
 - a. At 30 cm from the center of the permanent soil-sampling plot, mark the location for soil excavation at a cardinal direction following the order from north (1st carbon inventory) – east (2nd carbon inventory) – south (3rd carbon inventory) – west (4th carbon inventory); after four carbon inventories, soil excavation could be repeated at the old sampling locations following the above order.
2. At the soil pit location:
 - a. Excavate the soil by hand (where applicable) to depth at MCD, where MCD is computed in step A (sub-step A.2.).
 - b. At the exposed soil profile surface, delineate the soil horizons (where applicable) using standard soil survey techniques (Soil Survey Division Staff, 1993), and record the lower depth and thickness of each horizon.
 - c. For each soil horizon (where applicable) or each 5-cm-thick soil layer to a depth at MCD, excavate by hand (where applicable) additional soil volume at minimum horizontal dimensions of 30 x 30 cm. Place all samples in heavy-duty, soil sampling bags, mark the appropriate soil horizon or depth on the bags, and transport to a laboratory for analysis.
 - d. For each sampled soil horizon or a 5-cm-thick soil layer, determine the volume of the excavated soil using the displacement method by lining the inside of the excavated hole with a thin plastic sheet and filling it with steel BB's. The BB's are transferred to a graduated cylinder to determine their volume in cm³, which is recorded as the volume of the excavated soil per soil horizon or a 5-cm-thick soil layer.
3. Laboratory analysis:
 - a. The soil samples (from 2c) are air-dried and the total soil weight is recorded. The entire soil volume is passed through a 2-mm mesh sieve to separate the fine (< 2 mm) and coarse (> 2 mm) fractions of the soil, and their respective weights are recorded.
 - b. Bulk density and the volumetric content of the fine soil fraction for each soil horizon (or each 5-cm-thick soil layer) are determined as described in step A (sub-step A.2.) using the soil volume (from 2d) and weight estimates (from 3a).
 - c. Soil carbon concentration as percent by sample weight is determined in the fine soil fraction using the 16-step SOC estimation method (Substudy 1).
 - d. If sufficient funds are available, the nutrient content in the fine soil fraction should also be measured for further site characterization.
4. Data analysis:
 - a. For each soil horizon (or each 5-cm-thick soil layer) determine the SOC(g m⁻²) content using Eq. 4. Use the estimated values for bulk density and the volumetric content of the

fines (from 3b), the carbon concentration of the fines (from 3c), and the horizon thickness (from 2b) to substitute for the respective parameters in Eq.4.

- b. The total SOC(g m^{-2}) content down to MCD depth is estimated as the sum of SOC contents within each soil horizon (or each 5-cm-thick soil layer).
- c. Estimate the mean and standard deviation of the total SOC(g m^{-2}) per stratum (from 4b) using the data from each sampling location within the stratum.
- d. For each stratum, determine the carbon credits from the soil carbon pool by subtracting the baseline SOC content from the mean SOC(g m^{-2}) (from 4c). For each stratum, determine the number of “significant” years until the next SOC inventory using the computational methods in step A (sub-step A.2.).

C.2. Vegetation biomass.

For each stratum, determine the carbon credits from the vegetation biomass carbon pool by subtracting the baseline vegetation biomass carbon content from the mean carbon content estimated from all plots within the stratum. The carbon content in the different components of the vegetation carbon pool in a forested ecosystem is estimated as follows:

C.2.1. Overstory canopy layer.

Diameter at breast height (DBH) is measured and recorded by tree species for all trees with DBH greater than 2.5 cm that are located within a 12-m-radius circular area (equal to 452 m^2) from the center of each vegetation sampling plot. Total tree biomass (including stem wood, stem bark, foliage, treetops, branches, stumps, and coarse roots) is estimated using biomass equations (Jenkins et al. 2003). Total tree biomass is converted to kilograms of carbon using conversion ratios for different tree species groups (Birdsey 1992). The total tree biomass carbon content per plot, computed as the sum for all measured trees within the 452-m^2 area, is divided by the plot area to estimate the total tree biomass carbon content per unit area (Mg ha^{-1}).

C.2.2. Understory canopy layer:

At randomly chosen locations within each overstory measurement plot, establish four 2-m-radius circular sub-sampling plots (equal to 12.5 m^2 each) to collect samples of the understory vegetation. The four plots are randomly distributed so that each of the four quadrants of the overstory vegetation sampling plot is sampled.

In order to obtain a good estimate of the carbon sequestered and stored in the understory vegetation biomass, all tree saplings that have $\text{DBH} < 2.5\text{cm}$, or are less than 1.3 m tall, as well as all shrubs, which fall within, or originate from any of the four sub-sampling plot are clipped and the wet weight of the fallen biomass is recorded. The sum of the biomass weights from all four plots is also estimated (total biomass wet weight, kg)). A composite sample from all four plots is collected and brought to a laboratory for biomass moisture and carbon content determination. The composite biomass sample is dried at 65°C , and moisture content is estimated as $\text{Moisture}\% = [\text{wet biomass weight} - \text{dry biomass weight (kg)}] / [\text{dry weight (kg)}] * 100$.

Five biomass sub-samples, about 5 g each, are collected from the dry composite sample to be ground and passed through a 1-mm mesh sieve. The carbon content ($C_{\text{biom,wt}\%}$) of each sub-sample is measured using an automated carbon analyzer. The average of the five $C_{\text{biom}}(\text{wt}\%)$

measurements is used to measure the carbon content in Mg ha^{-1} (or g m^{-2}) units as follows:
$$C_{\text{biom}}(\text{Mg ha}^{-1}) = (\text{total biomass wet weight (kg)}) * C_{\text{biom}}(\text{wt\%}) / 100 * (1 / 50.27 \text{ m}^2) * (1 + \text{Moisture\%} / 100) * 10$$
, where 50.27 m^2 is the cumulative area of the four sub-sampling plots; and 10 is a unit conversion factor, which is used to transform biomass content from kg m^{-2} to Mg ha^{-1} units.

C.2.3. Grasses and legumes:

Use sampling procedures described in step B (sub-step B.2.).

C.3. Litter layer.

Use sampling procedures described in step B (sub-step B.3.).

Step D: Reference areas.

Determine the ecosystem carbon content in Mg ha^{-1} units within each reference stratum using the same sampling and analysis methods described in steps A, B, and C. Compute the net carbon sequestration increase due to mined site reforestation as the difference between the net ecosystem carbon stock of the reference sites (Mg C ha^{-1}) and the net ecosystem carbon stock of the treated sites (Mg C ha^{-1}). Similarly, if different forest establishment treatments are used across the project site, then estimate the net carbon sequestration increase for each treatment.

Site Selection, Characterization, and Site Quality Mapping Procedures

A/ Site selection: The criteria for selecting a suitable surface mined site for a carbon sequestration pilot project were as follows: (1) the site would be representative of all reclaimed mined lands in the hard-rock region of the Appalachian coalfield, in terms of the chemical and physical properties of the mine soil, with emphasis on the Clinch and Powell River valleys; (2) the site would potentially sustain good tree growth (hardwoods); (3) the topography and location of the site would allow at least 40 or more contiguous hectares ($>100 \text{ ac}$) of mined land to be reforested and managed; (4) the site would be of hayland/ pasture post mining land-use category; and (5) the site would be easily accessible via existing road network.

B/ Site characterization: At representative locations on the three best selected sites, we collected surface soil samples (0-20 cm) to evaluate the mine soil's chemical and physical properties. All soil samples were collected in paper or plastic bags, air-dried, passed through a 2-mm sieve and the content of the fine ($<2 \text{ mm}$ soil particles) and coarse fractions ($>2 \text{ mm}$) of the soil were measured and recorded along with other chemical and physical soil properties. Subsamples of the fine soil fraction were analyzed for plant-available nutrients at the soil testing lab at Virginia Tech. Results from these analyses were used to identify the one best site suitable for reforestation and tree growth.

C/ Site quality mapping: We used the site quality mapping procedures described in step A (sub-step A.1.) of the proposed field sampling and measurement protocol for carbon inventory on mined land. In addition to the methods described in the sampling protocol, we also used the most recent remote sensing (RS) data suitable for analysis in GIS to aid in the characterization and mapping of the project site. The boundaries of the project site were digitized on-screen using high resolution (1-m ground resolution), off-leaf condition aerial photos, available from the Virginia Base Mapping Program (VBMP, 2002).

Along with the high-resolution, true color imagery, the Virginia Base Mapping Program

(VBMP) also provided points with elevation data, which were used to rectify the aerial imagery acquired for this program. We determined that the point elevation data set could also be used to create digital elevation models (DEMs) for the project site. We used the *Geostatistical Analyst* extension in ArcGIS software (ESRI Inc., Redlands, CA 92373) to create the DEMs and elevation contour data sets for the project site. In a preliminary GIS analysis of the VBMP point data, we determined that the vertical accuracy of a predicted DEM was 2.4 m (8 ft) created at a 0.6-m (2-ft) horizontal resolution. The DEM was then used to generate aspect, slope, and curvature surface models as well as elevation contours for the extent of the project site. The ArcGIS software-integrated functions were used to generate all of the above surface models.

We surveyed the project site to identify and delineate the boundaries of all mapping units on the landscape within which soil chemical and physical properties were expected to be uniform. At representative locations across the mined landscape, we measured eight soil chemical and physical properties, along with two landscape properties at 47 point-sampling locations. All field measurements were made in October 2004. We overlaid the sample locations with the VBMP aerial photography to digitize (on-screen) in more detail the boundaries of the plantable area of the project site.

We used standard methods to measure soil pH, EC, soil texture, soil color, aspect, and slope on the selected project site (Soil Survey Division Staff, 1993). Detailed description of the field methods and techniques used to measure percent rock fragments (RF), percent sandstone (SS), soil density (compaction), and rooting depth are described by Jones (2005). For example, we used a screw auger to determine the maximum rooting depth based on the depth that was reached by screwing a 1-m long screw auger into the mine soil by an average-size person (about 84 kg or 185 lbs body weight) until the auger reached hard bedrock, or was obstructed by a dense subsoil layer. The average of three rooting depth measurements was used to determine soil depth. Based on the soil sampling location and soil property data, we reported and mapped areas of potential root growth limiting factors such as high soil density (c), insufficient soil depth (d), very high/low pH (p+/-), high rock fragment content (r), and unsuitable soil texture (t). The above designations for potential root growth limiting factors were adopted from Jones (2005).

The results from these field measurements were used for PI estimation, and eventually for FSQC calculation and mapping. We used the white pine-specific PI model described by Jones (2005) to map the prototype project site into strata of similar FSQC. We substituted the following importance factors for the texture, density and soil depth parameters in Eq. 2: $IF_{\text{txt}} = 0.21$, $IF_{\text{Db}} = 0.32$, and $IF_{\text{d}} = 0.47$, respectively (Jones 2005, p.103). We also mapped landscape features such as location and length of road features and drainage ditches for a more complete site characterization of the project site. All data from the field reconnaissance trips were entered in a GIS for further spatial analysis, and to create future maps of carbon content at different spatial scales.

Soil Sampling

Currently, because of the initial stages of the prototype carbon project, we did not estimate the sampling intensity for carbon inventory in the soil using the sampling procedures described in step A (sub-step A.2.) of the proposed protocol. Instead, for this paper, we used a low-cost modification of these procedures to produce a preliminary estimate of the sampling intensity necessary for SOC analysis on the project site.

We collected surface 0-20 cm soil samples from randomly selected locations across the landscape. The soil samples were air-dried and sieved through a 2-mm mesh sieve to separate the fine and coarse soil fractions. Using an automated carbon analyzer, we measured the total soil carbon content (C_{total}) as percent by sample weight, which included the carbon derived from soil carbonates, coal, and SOC; $C_{\text{total}} = C_{\text{carbonates}} + C_{\text{coal}} + \text{SOC}$. With estimates of the mean and standard deviation of C_{total} in Eq.3, we computed the preliminary sampling intensity necessary for carbon inventory in the mine soil for the prototype project site.

Vegetation and Litter Layer Biomass Sampling

We collected vegetation and litter layer biomass samples at the prototype project site using the sampling procedures described in step A (sub-step A.3.) of the proposed protocol. All biomass samples were collected from randomly chosen locations within the site and their coordinates were recorded with a GPS unit. The latitude and longitude coordinates of the sampling locations were used for further analysis and to produce maps.

Using the dry sample weights, carbon content measurements, and the precise sample-clipping area, we calculated $C_{\text{biom}}(\text{g m}^{-2})$ using Eq. 5. With estimates for the mean and standard deviation of $C_{\text{biom}}(\text{g m}^{-2})$ in Eq. 3, we computed the respective sampling intensities necessary for carbon inventory in vegetation and litter layer biomass on the prototype project site.

Results and Discussion

The proposed field sampling and measurement protocol for carbon inventory on mined land was evaluated for use in site characterization, site quality mapping, and sampling intensity estimation for ecosystem carbon analysis of a prototype carbon sequestration project on a mined land in southwestern Virginia. Supported by a group of governmental, non-profit, and research institutions, namely, The Nature Conservancy (TNC), the Department of Mines, Minerals, and Energy (DMME), and the Department of Forestry at Virginia Tech (DOFVT), the prototype project was aimed to demonstrate the potential for carbon sequestration of reforested coal mined lands located in the Clinch and the Powell River Valleys of southwestern Virginia. An appropriate mix of native hardwood tree species will be used to restore the forests that once grew on these lands.

Among many direct objectives of the prototype carbon project, investigators intend to use this project as a reforestation-marketing tool for mined land reclamation, and show that forest establishment on mined land for carbon sequestration projects could be an attractive post mining reclamation practice for future mined land reclamation projects in the Clinch and Powell River valleys. Therefore, careful site selection, comprehensive site characterization, and site quality mapping were important steps for this project.

Site Selection Results

It was very important to select a mined site that would represent the majority of surface-mined sites in the Appalachian coalfield. Not only would this allow the TNC, DMME, and DOFVT personnel to better understand the processes and interactions between different types of mine spoil materials and tree growth and survival, but it would also allow researchers to gain valuable knowledge about the carbon sequestration potential of mine lands of different soil quality and varying mine spoil properties. At the end, the findings from this prototype carbon sequestration

project could be used and transferred to any mined land in the Appalachian region to establish future forests for terrestrial carbon sequestration.

Seventy-six surface coal mined sites were identified as potential project sites using remote sensing data analysis such as Landsat TM satellite imagery analysis, aerial photography from the Virginia Base Mapping Program (VBMP, 2002), and field reconnaissance (in summer 2004). Most mined sites ranged from shrubland to unmanaged forests of poor quality. For example, 11 mined sites that were visited during the field trips were unmanaged forest sites and were unsuitable for the establishment of a carbon sequestration project based on the high initial costs for site preparation. Other sites that were also regarded as unsuitable for the establishment of a carbon sequestration project were either too steep or were inaccessible by vehicle due to the poor condition of abandoned roads and the presence of thick shrub vegetation.

The top three selected potential project sites were *Flint Gap 1* (latitude and longitude coordinates of 36°59'11"N and 82°15'20"W, respectively), *Weaver Creek Headwaters 1* (latitude and longitude coordinates of 37°2'7"N and 82°6'0"W, respectively), and *Stallard Cemetery/Possum Hollow 1* (latitude and longitude coordinates of 36°53'40"N and 82°23'40"W, respectively). The *Flint Gap 1* site was characterized with a mixed spoil type, mostly acidic in pH, and with diverse vegetation distribution across the landscape. In addition to the spoil diversity and accessibility of this site, the presence of many trees growing across the entire landscape implied an excellent show-and-tell opportunity if this site were selected as the prototype project site.

The *Weaver Creek Headwaters 1* was characterized with a vast and mostly uniform brown sandstone spoil type of low spoil compaction which implied excellent root growth media if the site were fertilized. This site was characterized with steep slopes, ranging from 12-18 degrees that could potentially limit the land area suitable for reforestation. If the *Weaver Creek Headwaters 1* site were selected as the prototype project site, a potential disadvantage would be that the site was owned by multiple owners who might prohibit transfer of land ownership in the future.

The *Stallard Cemetery/Possum Hollow 1* mine site differed from the other two potential sites. This site was mostly contour-mined and a significant fraction of its land area was present as high-wall and bench topographic land features. Although the shale spoil type guaranteed long-term fertility, the physical properties of the soil associated with root growth were expected to deteriorate in the future as the soil texture could transform from the moderate loamy to a finer texture such as loamy-clay or clayey. Fine soil textures are unsuitable for tree growth, and if not managed properly, there could be great erosion potential on large sections of the site as the vegetation cover could be compromised.

Site Characterization Results

In general, we found that the three best potential sites would be suitable for tree growth. The acidity (pH) values ranged from 4.2 to 8.0 for the three sites, and EC ranged from 0.01 to 0.12 mmhos cm⁻¹ (Table 1-3-1). Soil acidity and EC levels of the spoils on the three sites were in acceptable ranges suitable for growth of hardwood tree species (Rodrigue and Burger, 2004).

Table 1-3-1. Physical and chemical properties of mine soil samples collected from the three best potential carbon project sites in southwestern Virginia.

Parameter ID	#	<u>Stallard Cemetary 1</u>					<u>Weaver Creek 1</u>
		P21 1	P22 2	P23 3	P24 4	P31 5	T01 6
pH	1	5.5	4.5	3.3	5.2	6.2	5.5
EC, mmhos/cm	3	0.04	0.05	0.46	0.12	0.01	0.01
Texture	5	L	L	L	L	L	SL
CFC (%wt)	6	67	46	64	47	43	49
Depth (bot, cm)	2	20	25	5	20	20	15
Rock type	4	1	1	1	2	1	5
Macro nutrients, ppm in soil (mg/Kg)							
P	7	33.0	2.5	2.0	41.5	88.5	6.0
K	8	111.5	84.0	27.0	118.5	49.0	41.5
Ca	9	891.5	556.5	82.5	979.0	968.0	303.0
Mg	10	336.5	235.5	118.5	361.5	416.0	113.0
Micro nutrients, ppm							
Zn	11	3.4	2.3	1.9	3.5	2.0	1.7
Mn	12	16.9	13.2	9.2	19.3	5.9	13.6
Cu	13	1.0	0.9	2.1	0.8	2.5	1.1
Fe	14	72.1	14.4	48.8	37.2	26.2	20.4
B	15	0.2	0.2	0.1	0.4	0.1	0.1
Lime, Tons/Ac	17	--	3	N/A	2	--	--
Comments	16		Good white pine growth at edge of outslope; Located above gully erosion site	Composite sample: Gully erosion site		Stallard Cem 2 sample to characterize soil from bottom of highwall	--

Table 1-3-1 (continued).

Parameter ID		Flint Gap 1								
		T21	T22	T23	T24	T25	T26	T27	T28	T29
	#	7	8	9	10	11	12	13	14	15
pH	1	8.0	5.4	5.7	4.2	7.8	7.4	5.8	5.0	5.4
EC, mmhos/cm	3	0.07	0.02	0.01	0.05	0.05	0.05	0.01	0.03	0.01
Texture	5	L	L	L	SiL	L	L	L	L	L
CFC (%wt)	6	63	47	55	59	61	53	50	22	51
Depth (bot, cm)	2	15	25	25	25	15	15	15	20	20
Rock type	4	1	1 + coal	3	1	1	5	1	5	1 + coal
Macro nutrients, ppm in soil (mg/Kg)										
P	7	124.0	50.0	9.5	17.5	106.0	37.0	41.0	2.5	34.0
K	8	37.0	68.5	89.5	52.0	28.5	50.5	58.5	48.0	116.0
Ca	9	1690.0	493.5	351.0	233.5	1637.5	1085.0	486.0	163.5	495.5
Mg	10	440.5	223.0	185.5	168.0	432.0	271.0	146.5	59.0	222.0
Micro nutrients, ppm										
Zn	11	4.5	5.1	1.4	3.5	3.3	2.4	1.3	2.1	5.9
Mn	12	53.9	13.4	7.7	11.9	46.1	44.7	7.2	13.0	14.1
Cu	13	2.5	4.3	1.7	2.3	1.7	2.1	2.8	2.4	4.5
Fe	14	42.0	51.1	35.8	23.9	34.5	46.8	18.6	27.3	37.4
B	15	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Lime, Tons/Ac	17	--	2	1	4	--	--	1	--	--
Comments	16		Sourwood patch						Sourwood patch	Sourwood patch
		--		--	--	--	--	--		

Notes: A.Trees (apples) are expected to respond to fertilizer application for Low and Medium testing categories:

Low is indicated as dark gray; Medium - light gray. B.All micronutrients are in sufficient amounts.

Abbreviations: L = loam; SiL = silt loam; SL = sand loam; EC = electrical conductivity;

Rock Type: (1)100%SiS;(2) 75%SiS:25%SS;(3) 50SiS:50SS;(4) 25SiS:75SS;(5) 100%SS. SiS=siltstone;SS=sandstone

The soil texture was predominantly loam, indicating that there was a good soil-air-water balance at the current stage of mine soil genesis on all three sites. The sufficiency of micro- and macronutrients was positively related to the relative amount of siltstone in the mine soil (Table 1-3-1). Siltstone and certain types of shale are rich in nutrients and have excellent chemical properties that are critical for good tree growth. However, it is common for siltstone- and shale-derived mine soils to compact to greater densities as a result of machine traffic that could significantly inhibit tree growth (Burger and Zipper 2002). Because siltstone and shale weather to silt- and clay-size ($<0.05\text{mm}$) soil particles, with time, the soil texture could shift to a finer texture class (silt or silt-clay) and create poor soil-air-water balance in the rhizosphere of tree roots.

Therefore, we used rock type composition of the surface 30 cm layer of the mine soil as a predictor for tree-growth suitability on mined lands. Based on our observations, we determined that a balanced mixture between weathered siltstone (SiS) and weathered sandstone (SS) in the mine soil (ranging from 75/25 to 25/75 SiS/SS ratio) be the underlying criterion for choosing future project sites suitable for tree growth. A mixture of 75/25 sandstone-to-siltstone ratio would result in a sandy-loam texture that would be ideal for tree growth (Burger and Zipper, 2002).

The results from our preliminary soil sampling showed that the *Flint Gap 1* site (referred to as the *Flint Gap* site in the remaining sections of this text) had the most desirable rock type mixture. In contrast, the *Stallard Cemetery/Possum Hollow 1* site was on the “siltstone” end of the rock-type spectrum and the *Weaver Creek Headwaters 1* site was on the “sandstone” end of that spectrum (Table 1-3-1). Furthermore, based on the variety of rock-types present on the *Flint Gap* site (Table 1-3-1), unlike the *Stallard Cemetery/Possum Hollow 1* and the *Weaver Creek Headwaters 1* sites, the *Flint Gap* site would most likely represent the majority of all post surface coal-mining reclaimed sites in the hard-rock region of the Appalachian coalfield, and the Clinch and Powell River Valleys in particular. That is, the effects of different forest establishment treatments designed to maximize carbon sequestration on a variety of mine spoil types present on the *Flint Gap* site could be expanded to the majority of mine lands located in the Clinch and Powell River valleys for future carbon sequestration projects. Therefore, the *Flint Gap* site was chosen the site for a prototype carbon sequestration project.

The Flint Gap mine site was located in western Russell County, Virginia (Figure 1-3-2). The site was mined prior to 1996 and all mining operations were completed by February 14, 1996, according to the mining completion letter (Brad Kreps of TNC, pers. comm.). Site reclamation, tree planting on the slopes (mostly white and Virginia pines), and grasses/legumes seeding on the flat areas of the Flint Gap site, were all completed by March 17, 1997.

Detailed and verifiable information about previous land use practices was not readily available for the Flint Gap site. By word of mouth, we discovered that between 1997 and 2004 this site was leased to farmers for cattle grazing during the summer months, and that hunting leases had been provided to members of local hunting clubs who often had been visiting the site (and still did in summer 2005) for hunting-spots maintenance and possibly supplying food and forage for the wildlife.

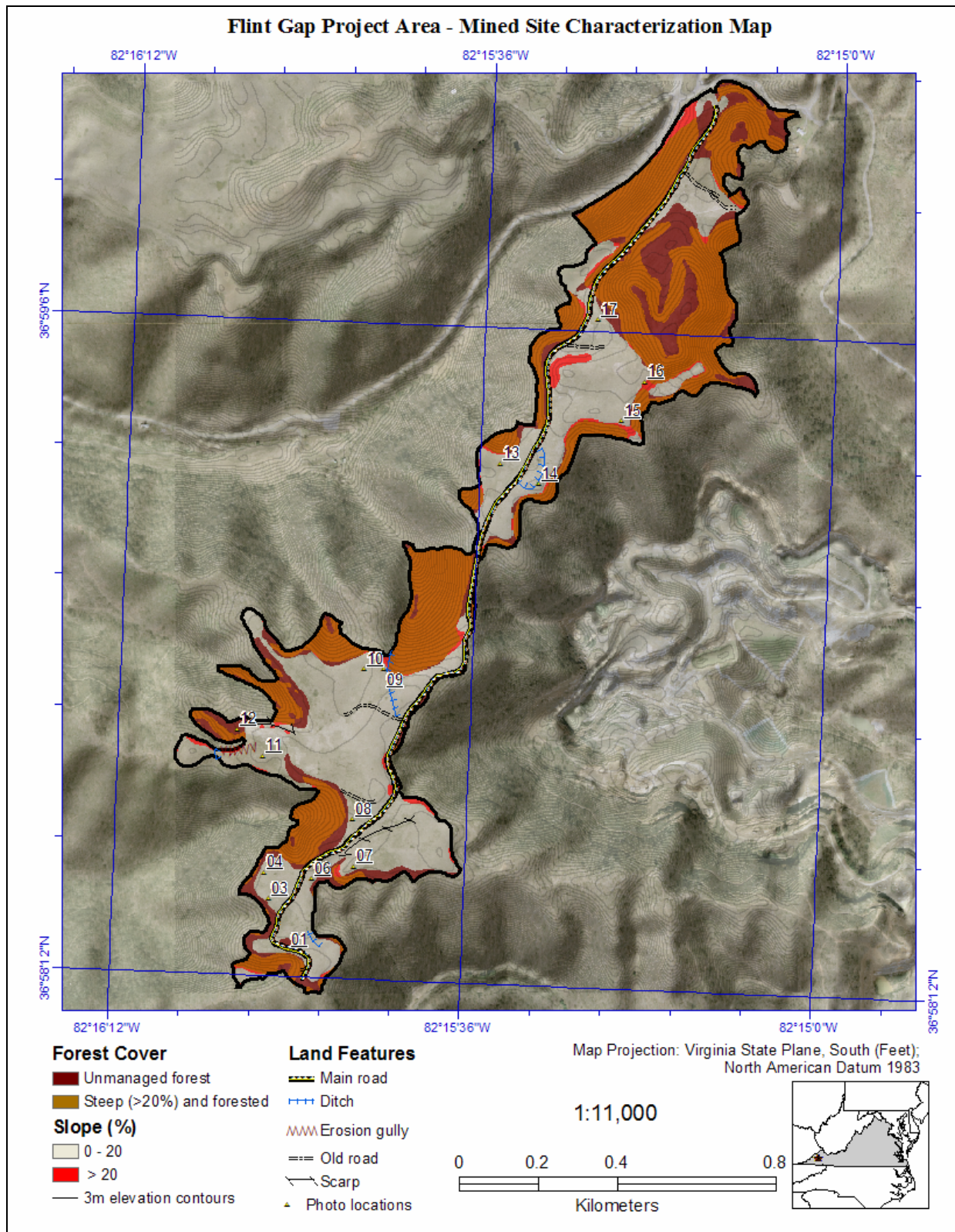


Figure 1-3-2. Project site boundaries, landscape features, forest cover, and plantable area of the Flint Gap mined site in Russell County, VA. Numbers on map indicate the location and the identification of reference photographs taken on the site in October 2004.

We digitized the boundaries of the Flint Gap site on-screen using high-resolution (1-m ground resolution), off-leaf-condition aerial photos available from the Virginia Base Mapping Program. In October 2004, the digital maps created from remote sensing data for the Flint Gap project area were verified in the field, and data for additional landscape features such as drainage ditches, old roads, scarps, and erosion gullies were collected and added to the existing maps (Figure 1-3-2). During this field surveying trip, sufficient soils data were collected to determine and map the site quality of the project area.

The entire land area of the Flint Gap site was easily accessible by vehicle despite the narrow and elongated shape of the site extending for more than 2.5 km from the southernmost to the northernmost corners. There was a well-maintained gravel road 2.7 km in length that followed the shape of the site and allowed ready access to any area on this project site (Figure 1-3-2).

The total surface area of the Flint Gap site was 75 ha, of which 35 ha were considered plantable and suitable for tree growth. The two main factors affecting the area's suitability for planting and tree growth were (1) current vegetation cover (areas covered with grass/legume vegetation were preferred as opposed to thick shrub patches), and (2) heavy equipment accessibility, which was affected by the hillsides' steepness; slopes greater than 20% were considered too steep to be traversed by machines used to establish, manage, and harvest the future forests, therefore, such areas were deemed unsuitable for reforestation for this project.

Thirty ha of the Flint Gap project area had slopes greater than 20% and were entirely covered with dense shrub vegetation or forest vegetation of poor tree quality. About 10 ha of the < 20%-sloped land on Flint Gap was also forested. Although most of the planted trees (Virginia pine and black locust) had survived for more than eight years following site reclamation in 1997, because of improper site preparation, mainly the lack of soil tillage and fertilization, the overall tree health did not appear satisfactory for timber production (Figure 1-3-2). In order to correct this problem and make up to the objectives of this project, many resources would be spent to remove the current forest vegetation and dense shrub patches (autumn olive shrubs) to properly prepare the site for tree planting, and re-plant desirable tree species that would grow into a healthy native forest ecosystem.

On isolated locations across the Flint Gap site, there was evidence for severe erosion potential, moderate to high compaction problem, and standing water and wetland vegetation. The field-surveying data showed that a total of 2.8 ha of the plantable areas on the site had severe erosion potential; the most significant erosion problem was observed to the east of the locations of photos 11 and 12 in Figure 1-3-2. The soil acidity level (pH of 3.5) at these locations were very high indicating that only sparse grass or tree vegetation could survive and form vegetation cover to protect the bare soil against erosion.

There were two distinct areas on the site with combined total area of 0.8 ha that could be identified as emergent wetlands due to the observed standing water and wetland vegetation. The centers of these two wetland areas were located at the respective latitude and longitude coordinates: (1) 36°58'16.9"N and 82°15'53.0"W; and (2) 36°58'33"N and 82°15'51.8"W. From a forestry perspective, growing trees on wetland areas is possible, but the only mechanism to help the tree seedlings survive the initial establishment would be through bedding. Ultimately, it would be the landowner's decision as to whether to protect these future wetland areas or to grow native tree species.

Approximately 2 ha of the plantable area on Flint Gap were considered to be moderately to highly compacted. Five of the seven compacted zones were located to the east and south of the location of photo 6 in Figure 1-3-2. Because soil compaction is one of the most critical soil physical properties affecting root and tree growth, sites of high mine soil compaction have low FSQC, usually class III or lower; on such sites, trees grow very poorly if they survive the initial years. Broadcast soil tillage to about 1-m depth has been shown to alleviate soil compaction and improve tree growth (Ashby, 1997). Tillage would improve the soil-air-water balance on compacted sites and could ultimately result in a permanent increase in site quality to a desirable FSQC of I.

Site Quality Mapping Results

Our initial approach was to sample the site at regular intervals across the landscape, following a random systematic sampling approach (Dollhopf 2000) by pushing a sharp-edged shovel into the surface soil layer (down to 0-15 (20) cm depth) and look for differences and changes of soil color, soil compaction, content of coarse rock fragments, and observing any changes in the vegetation type and vigor. However, we soon noticed that the boundaries of the mapping strata that were delineated by intensive soil surveying as described above could also be delineated on the VBMP imagery by simply drawing out the areas of different color and color saturation across the site. For example, wet patches of land such as emerging wetland areas appeared as dark, nearly black spots on the VBMP aerial imagery due to the higher absorbance of visible light by water and wet soils. In comparison, dry and particularly rocky and shallow sections on the project site appeared as light, almost white patches of land.

Using the combined information from the VBMP aerial imagery, the changes in vegetation type and vigor observed in the field, and by testing the surface soil for change in the easily-detectable soil properties such as relative soil compaction, soil color, and relative coarse rock content, we identified and delineated 51 continuous polygons (strata) of uniform FSQC (Figure 1-3-3). These polygons were assigned the estimated FSQC class of the sampling locations (Jones 2005) based on the soil properties measured at all sampling locations within each stratum. For seven polygons, we used the FSQC class estimated for the surrounding bigger area because of similarities for all but one of the measured soil properties. For example, the smaller polygon in one case was slightly less compacted relative to the surrounding bigger area, or the smaller polygon for another case had slightly higher coarse rock fragment content relative to the surrounding area. From a practical point of view, because of the smaller size of these seven polygons, they would be managed the same way as the surrounding bigger area of similar FSQC class.

On average, about 70% of the plantable area on Flint Gap was of acceptable site quality. The results showed that 40% of the total plantable area on the Flint Gap site was of FSQC I, followed by FSQC II (32%), and FSQC III (27%) (Figure 1-3-3). There was only one polygon (0.5 ha) of FSQC IV that represented about 2% of the total potential area for reforestation on Flint Gap (Figure 1-3-3).

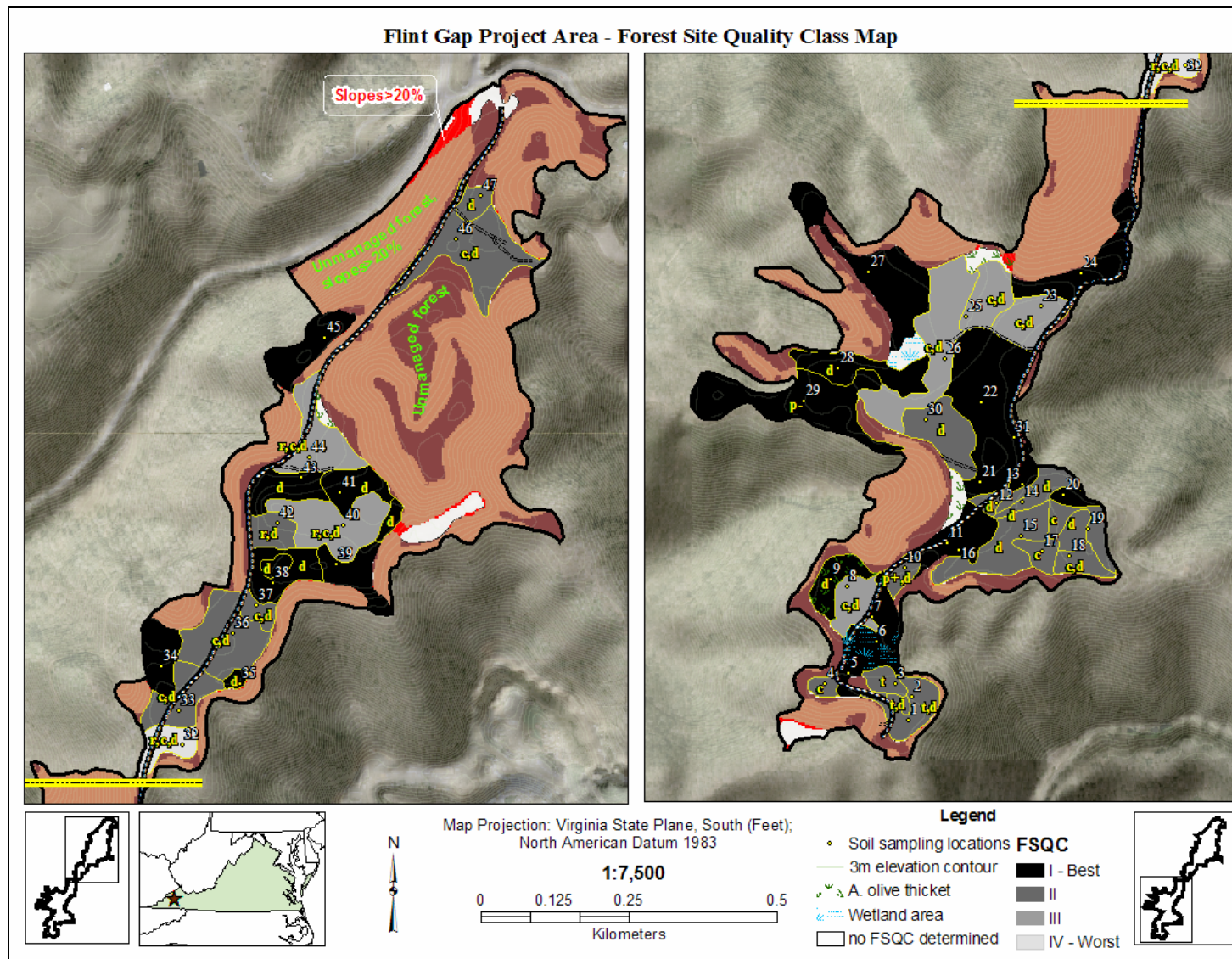


Figure 1-3-3. Forest site quality class polygons delineated across the Flint Gap project site located in Russell County, Virginia. Letter symbols indicate the specific soil limitation(s) for tree root growth identified at each sampling location, and numbers indicate the *ID* number of the sampling location (Table 1-3-2).

Based on the FSQC of each stratum on Flint Gap, the monetary profits from wood production (mine props, pulp, small and large sawtimber) could vary by one order of magnitude across the site (Burger and Zipper 2002). Burger and Zipper (2002) reported that the profits from white pine stand management on mined land of FSQC IV would be about \$ 300 ha⁻¹ mainly from mine props and pulp production, while the profits from mined lands of FSQC II would be nearly \$8,600 ha⁻¹ from large sawtimber production. The return on investment (%) from reforestation on mined land increases exponentially with the increase of FSQC as follows: -5 on FSQC V, 0 on FSQC IV; 3 on FSQC III; 7 on FSQC II; and 10 on FSQC I (Burger 2002).

Today, there is sufficient evidence from mined land research studies (Rodrigue 2001; Rodrigue and Burger 2004; Rodrigue et al. 2002), and existing practical guidelines (Burger and Zipper 2002) for reforestation of reclaimed mined land that could be used to establish productive forests, which would generate additional profits for mined land owners. Additionally, reforested mined land would also sequester atmospheric CO₂ in the mine soil and in wood products, especially in wood products that stay in use for many years, often greater than 100 (Birdsey and Lewis 2003; Miner 2003; Skog and Nicholson 1998; Smith et al. 2006), which would help lower the greenhouse gas (GHG) emission levels for the United States and meet the designated GHG caps endorsed in the Kyoto protocol (COP-UNFCCC 1997).

In order to maximize the profits from wood production and reach the maximum carbon sequestration potential of the land, some areas in Flint Gap would require more specific site preparation treatments, such as soil tillage. The mine soil properties for some of the strata on Flint Gap indicated that certain root growth limiting conditions could develop in the future (Figure 1-3-3 and Table 1-3-2). The mine soil of approximately 21 ha of the plantable area on Flint Gap was surveyed as either being too dense or too shallow for tree root growth (symbols *c* and *d* in Figure 1-3-3 and Table 1-3-2). This is usually a result of heavy machinery trafficking across the site during site reclamation, or using insufficient amount of topsoil or topsoil-substitute as a root growth medium (Burger 2002; Burger and Zipper 2002).

Excessive compaction of soils by trafficking heavy mining machinery could inhibit the survival and growth of planted trees (Ashby 1997; Torbert and Burger 1990; Torbert and Burger 1996; Zeleznik and Skousen 1996). Torbert and Burger (1990) reported significant decrease in tree survival on graded and tracked-in soils compared to rough-graded sites on a partially reclaimed mined land site in Wise County, Virginia. Percent survival ranged from 0% (sycamore) to 63% (Virginia pine) on graded sites, and from 42% (Virginia pine) to 100% (sugar maple and black walnut) on rough-graded sites (Torbert and Burger 1990). The height response ranged between 0% (red oak) and 100% (sugar maple) (Torbert and Burger 1990).

Ashby (1997) showed that soil tillage (ripping) on mined lands, in addition to competition control treatments, could significantly improve tree growth by alleviating soil compaction and increasing the volume of the rooting zone. The overall data analysis indicated that the combined effect of ripping and herbicide treatment significantly increased tree height growth (+84% versus control), compared to ripping-only (+42% versus control), herbicide-only (+20% versus control), and the control (no ripping and no-herbicide treatments) (Ashby 1997). Therefore, soil tillage could be used as one of the site preparation treatments on Flint Gap prior to tree planting on each stratum with potential soil density and rooting depth limitations for tree growth (Figure 1-3-3 and Table 1-3-2).

Table 1-3-2. Mine soil properties used to determine the PI_{wp} and FSQC (Jones, 2005) at 47 sampling location on the Flint Gap project site. Data were collected in October 2004, when the site was 7 years old.

Point ID	Texture	Density	Rooting depth (cm)	PI_{wp} *	FSQC 'I' to 'V'	Symbol^	Point ID	Texture	Density	Rooting depth (cm)	PI_{wp} *	FSQC 'I' to 'V'	Symbol^
1	SiL	low	45	0.75	II	t,d	25	L	moderate	36	0.65	III	c,d
2	SiL	low	46	0.75	II	p+,t,d	26	L	moderate	40	0.65	III	c,d
3	SiL	low	63	0.78	II	t	27	L	very low	80	0.95	I	
4	L	moderate	>90	0.71	II	c	28	SL	low	40	0.84	I	d
5	L	low	68	0.82	I		29	L	low	>100	0.85	I	p-
6	L	low	>75	0.85	I		30	L	low	40	0.79	II	d
7	L	low	>75	0.85	I		31	SL	low	60	0.87	I	
8	L	moderate	37	0.65	III	c,d	32	SL	high	10	0.43	IV	r,c,d
9	SL	low	33	0.81	I	d	33	SL	moderate	45	0.70	II	c,d
10	L	low	45	0.79	II	p+,d	34	SL	low	>75	0.91	I	
11	SL	low	60	0.87	I		35	SL	low	44	0.84	I	d
12	L	low	40	0.79	II	d	36	SL	moderate	41	0.70	II	c,d
13	L	very low	>75	0.95	I		37	SL	moderate	37	0.70	II	c,d
14	L	low	40	0.79	II	d	38	SL	low	45	0.84	I	d
15	L	low	45	0.79	II	d	39	SL	low	>100	0.91	I	
16	L	low	72	0.82	I		40	SL	moderate	21	0.64	III	r,c,d
17	SL	moderate	50	0.73	II	c	41	SL	low	38	0.84	I	d
18	SL	moderate	32	0.67	II	c,d	42	SL	low	17	0.75	II	r,d
19	SL	moderate	52	0.73	II	c	43	SL	low	42	0.84	I	d
20	SL	low	90	0.91	I		44	SL	moderate	24	0.64	III	r,c,d
21	L	low	68	0.82	I		45	L	low	>100	0.85	I	
22	L	low	52	0.82	I		46	SL	moderate	34	0.67	II	c,d
23	L	moderate	35	0.65	III	c,d	47	L	low	37	0.79	II	d
24	L	low	61	0.82	I								

* $PI_{wp} = (\text{texture} * IF_{\text{txt}}) + (\text{density} * IF_{\text{Db}}) + (\text{depth} * IF_{\text{d}})$, where of $IF_{\text{txt}} = 0.21$, $IF_{\text{Db}} = 0.32$, and $IF_{\text{d}} = 0.47$ were adopted from Jones (2005); SiL= silt loam; L=loam; SL=sand loam.

^ Ordination symbol indicates that sufficiency of soil property was ≤ 0.6 ; c=density, d=depth, p=pH (“+” for high, and “-” for low), r=rock fragments, t=texture;

~ Indicates slope of 0-1% or nearly level.

Approximately 7 ha of the plantable area on Flint Gap was mapped with potential root growth limitations due to low (or high) soil acidity, high rock fragment content, and fine soil texture designated with symbols p-/, r, and t, respectively in Figure 1-3-3 and Table 1-3-2. Burger and Zipper (2002) suggested that appropriate tree species should be selected for planting on areas with specific root growth limitations. For example, sugar maple and sycamore should be planted on more alkaline (pH between 6.5 and 8.5) and low compaction spoils, while Virginia pine and pitch x loblolly pine trees could be planted on more acid (pH between 4.5 and 6.5) and high compaction spoils (Burger and Zipper, 2002).

Soil Sampling Intensity on Prototype Carbon Project Site

The sampling intensity for baseline carbon estimation on Flint Gap was based on the range of C_{total} from preliminary soil samples collected from the site using a systematic soil sampling design. We collected a total of 162 surface soil samples (0-20 cm) on two randomly located 0.2-ha square plots (45 x 45 m). The only criteria for choosing the random sampling plots on Flint Gap was that a minimum number of sampling plots should include at least three of the four FSQC classes mapped on the site. For example, one of the sites included strata of FSQC I, while the other included strata of FSQC II and IV. The sampling intensity for strata of FSQC III was computed as the average for FSQC II and IV, assuming that the variation of C_{total} on strata of FSQC III was lower compared to strata of FSQC IV, and was higher compared to strata of FSQC II.

On average, the C_{total} was highest for the strata of FSQC II (2.73 wt%), followed by strata of FSQC IV (2.36 wt%), and FSQC I (1.94 wt%). Similarly, the standard deviation of these estimates was highest for strata of FSQC II (0.628 wt%), followed by FSQC I (0.512 wt%), and FSQC IV (0.506 wt%). Because these estimates included all pedogenic (plant-sequestered), geogenic (coal-derived), and inorganic (carbonates) carbon forms, no inferences could be made about the effect of FSQC on the carbon sequestration potential of the mine soil on the Flint Gap site.

The results from analysis of variance (ANOVA) of the C_{total} content measurements indicated that C_{total} for strata of different FSQC on Flint Gap were significantly different at the 0.05 alpha level; therefore, different sampling intensity levels for C_{total} measurement were computed for strata of FSQC classes I, II, and IV (Figure 1-3-4). For example, the sampling intensity for SOC measurement at the 90% precision and 20% accuracy levels on strata of FSQC I was 27 samples ha^{-1} (Figure 1-3-4). At the same accuracy and precision levels, the soil sampling intensities for strata of FSQC II and IV were 37 and 40, respectively.

Clearly, significantly higher soil sampling intensities were required to estimate SOC at higher accuracy (or precision) levels (Figure 1-3-4). For example, for strata of FSQC I, about 250% more sampling locations would be required to estimate SOC (at 90% precision) with 10% accuracy compared to 20% accuracy (Figure 1-3-4). In contrast, about 150% more samples would be required to estimate SOC (at 20% accuracy) with 99% precision compared to 90% precision. Ultimately, both the precision and accuracy levels for SOC analysis should be selected to meet the reporting standards for carbon credits in a future carbon market in the United States.

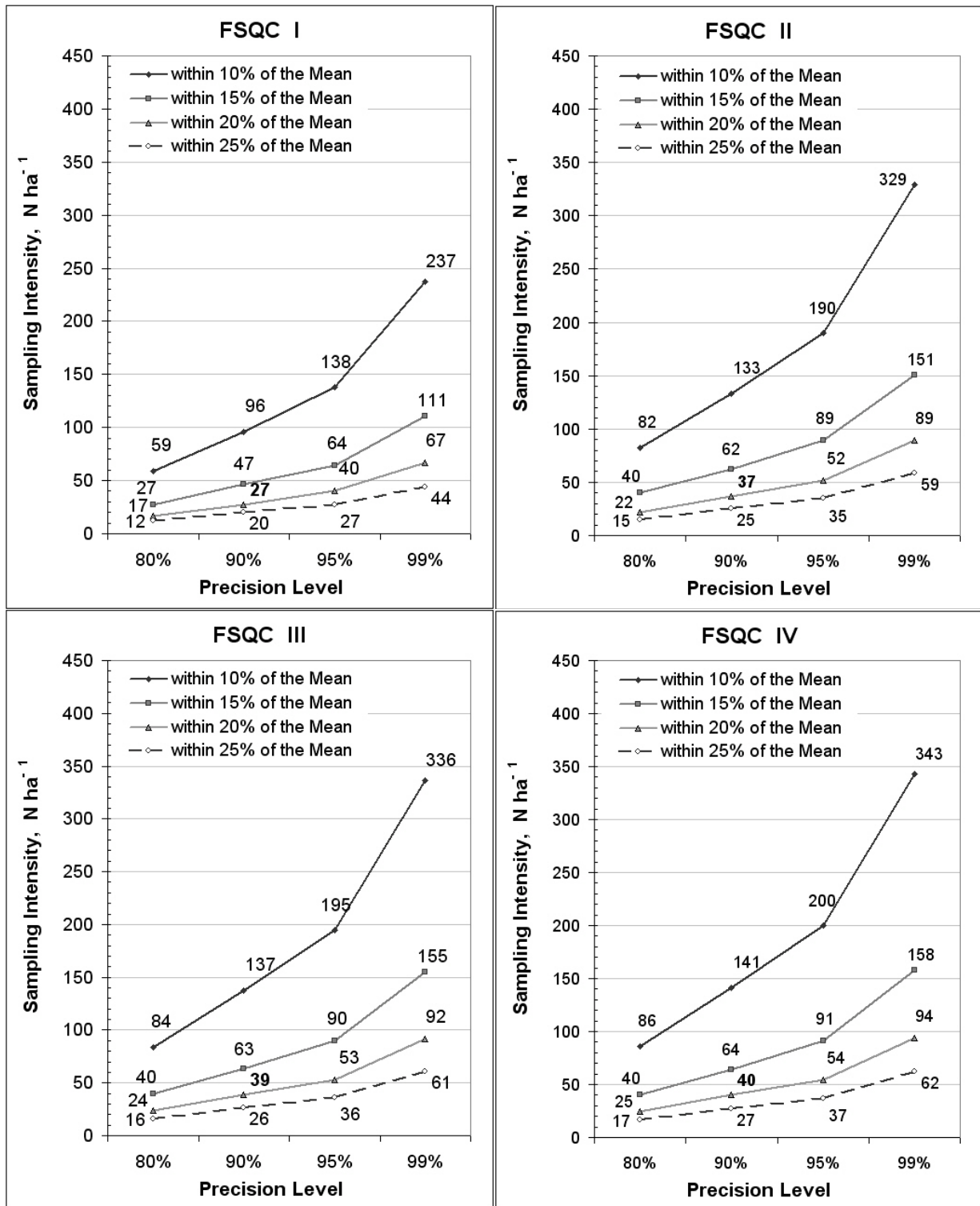


Figure 1-3-4. Sampling intensities necessary to measure C_{total} content on the Flint Gap site at four accuracy levels (10, 15, 20, and 25% from the mean) and four precision levels (80, 90, 95, and 99%). The four graphs represent the respective sampling intensities for strata of FSQC I-IV.

The sampling intensities presented in Figure 1-3-4 are based on the variation of C_{total} , which means that the number of samples required to measure SOC in the mine soil would be less. Assuming that the amounts of coal- and carbonate-derived carbon in a mine soil were insignificant, the variation of C_{total} would be equal to that of SOC. However, as mentioned above, C_{total} for most mine soils is a sum of the carbon content derived from soil carbonates, coal, and SOC. Hence, the standard deviation (St.Dev.) of C_{total} , based on the rules of propagation of uncertainty (Harris, 2005) would be estimated as $\text{St.Dev.}_{C_{\text{total}}} = \text{SQRT}((\text{St.Dev.}_{C_{\text{carbonates}}})^2 + (\text{St.Dev.}_{C_{\text{coal}}})^2 + (\text{St.Dev.}_{\text{SOC}})^2)$, where $\text{St.Dev.}_{C_{\text{carbonates}}}$, $\text{St.Dev.}_{C_{\text{coal}}}$, and $\text{St.Dev.}_{\text{SOC}}$ are the standard deviations of the carbon content in carbonates, coal, and SOC, respectively. Clearly, the $\text{St.Dev.}_{C_{\text{total}}}$ is larger than any of the above components, including $\text{St.Dev.}_{\text{SOC}}$.

Because soil sampling could be a very tedious and expensive task, especially on rocky and compacted mine soils, fewer resources would be spent if the SOC content was estimated by the sampling and measurement procedures described in the proposed protocol. An accurate and precise estimate of SOC made by the proposed methods in the protocol would have smaller variance compared to the variance of C_{total} ; hence, fewer soil samples would be required to measure SOC for carbon inventories.

Consider the sampling intensities for SOC estimation at the 90% precision and 20% accuracy levels (Figure 1-3-4); if these were the reporting standards for carbon credits in a future carbon market, then 27, 37, 39, and 40 soil sampling plots ha^{-1} would be necessary for measuring SOC on Flint Gap for strata of FSQC classes I, II, III, and IV, respectively (Figure 1-3-4). In order to ensure that sufficient number of sampling plots was present for future carbon inventories, at least 20% more plots should be located on each stratum (Figure 1-3-5). A simple computation would suggest that for a 0.88-ha stratum of FSQC I, there must be 29 ($0.88 \times 27 + 0.88 \times 27 \times 20\%$) sampling plots, including 20% reserve plots, to measure SOC at the above-selected precision and accuracy levels (Figure 1-3-5). To locate the plots on the ground as the centers of a square grid, as described in step A (sub-step A.4.) of the proposed sampling and measurement protocol, the spacing of the grid would be approximately 18.6 m estimated as $\text{SQRT}(10,000 \text{ m}^2/29)$ (Figure 1-3-5).

For future carbon inventories, before any soil samples are collected from the permanent soil sampling plots located on the Flint Gap site (Figure 1-3-5), the vertical distribution of sequestered SOC must be determined. It is possible that SOC analysis for carbon credits would be warranted to depths exceeding 100 cm (i.e., $\text{MCD} > 100 \text{ cm}$); however, the costs of SOC analysis may also indicate that such analysis may not be cost-effective (i.e., $\text{MCD} = 0 \text{ cm}$) (Substudy 2).

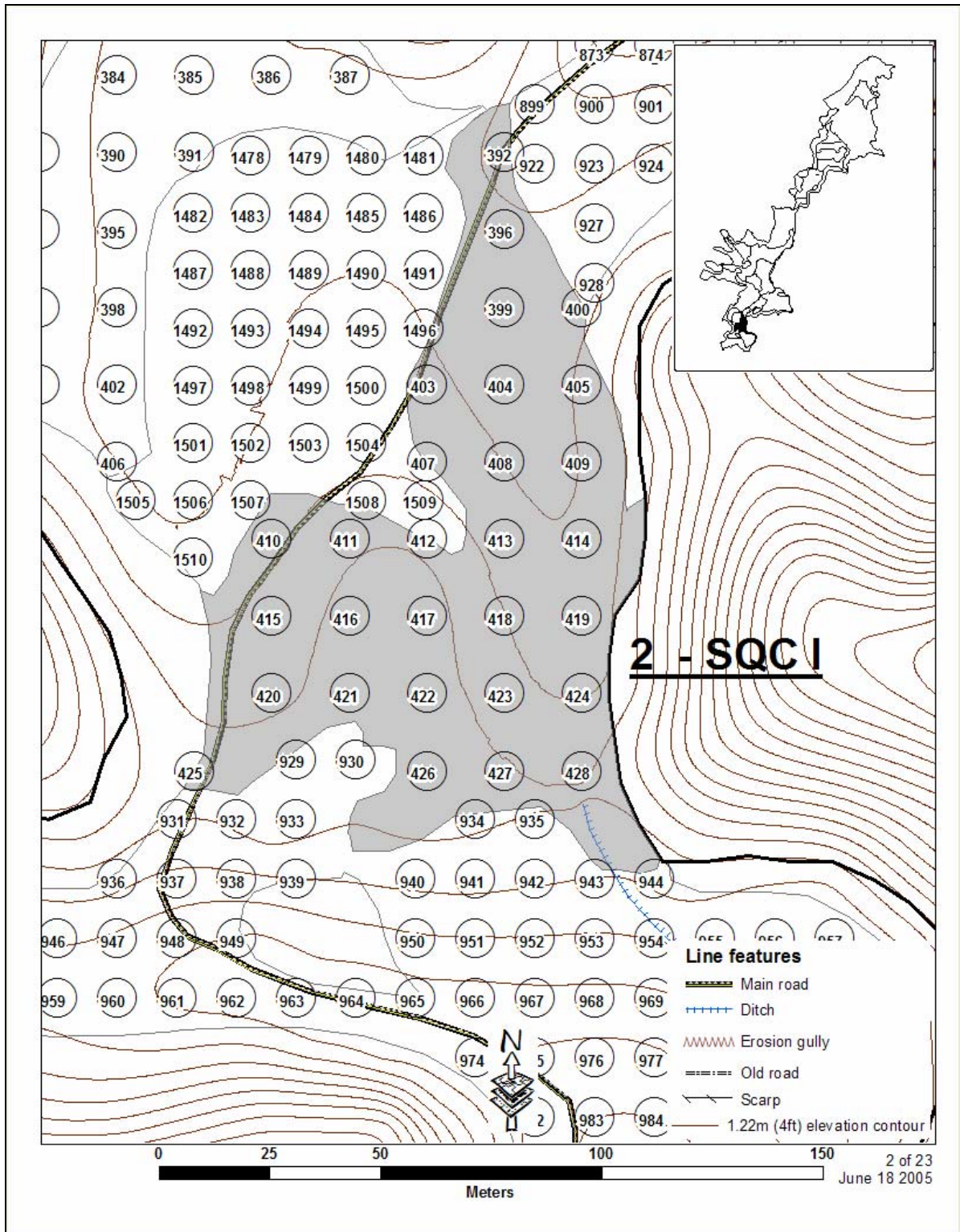


Figure 1-3-5. Permanent soil sampling plot locations for a mapping stratum (gray area; about 0.88 ha in size) of FSQC I on the Flint Gap project site. Sampling intensity of 27 plots ha⁻¹ was used to measure SOC at the 90% precision and 20% accuracy levels; an additional 20% plots (equal to 5) were marked as reserve.

Vegetation Sampling Intensity on Prototype Carbon Project Site

The flat areas of the Flint Gap site were seeded to a mixture of grass and legume species at the time of site reclamation. The grass species that were observed on the site were annual ryegrass (*Lolium multiflorum* Lam.), orchard grass (*Dactylis glomerata* L.) and timothy (*Phleum pratense* L.), while the legumes were red clover (*Trifolium pratense* L.), white clover (*Trifolium repens* L.), *Sericea lespedeza*, and birdsfoot trefoil (*Lotus corniculatus* L.). In a preliminary analysis, we collected vegetation (grasses and legumes) and litter layer biomass samples from randomly selected locations on the plantable areas at Flint Gap in order to estimate the variation of biomass carbon content across the landscape, and determine the number of permanent sampling plots for biomass carbon analysis.

Initially, biomass samples were collected from mapping strata that represented the areas of each identified FSQC class on the site, FSQC I through IV. However, for the strata of FSQC III and IV, because the vegetation and litter layer biomass samples were mixed together in the field, separate analyses were not possible. As a result, only biomass carbon analyses for strata of FSQC classes I and II were possible, and these data were used in the estimation of the respective sampling intensities for vegetation and litter layer carbon analyses on the Flint Gap site. The biomass samples from strata of FSQC I and II, a total of 429 samples, represented more than 70% of the plantable area on Flint Gap (Figure 1-3-3), which warranted the use of these carbon data in the estimation of biomass sampling intensities on the entire site.

We measured the carbon concentration of 23 and 22 vegetation and litter layer biomass samples, respectively. The results from an ANOVA analysis indicated that the carbon content of vegetation biomass measured as percent of sample weight (C_{biom} , wt%) was similar in both the standing vegetation and the litter layer biomass ($P = 0.5132$, $n = 45$). In contrast, C_{biom} (wt%) was significantly different for strata of different FSQC class ($P = 0.0033$, $n = 45$). On average, the C_{biom} (wt%) on strata of FSQC I was estimated at 47.64%, with standard error of the mean at 0.443%, and 49.04% for strata of FSQC II, with standard error of the mean at 0.224%. These average C_{biom} (wt%) estimates were used to estimate the amount of biomass carbon per unit area (C_{biom} , g m⁻²) for the rest of the vegetation and litter layer biomass samples, a total of 195 and 189 samples, respectively.

The results from an ANOVA analysis using the C_{biom} (g m⁻²) data for vegetation and litter layer indicated that the effect of FSQC on C_{biom} (g m⁻²) in the vegetation and litter layer biomass was not significant ($P = 0.1766$), while the effect of biomass sample type (vegetation versus litter layer) was significant ($P = 0.0005$). Prior to the data analysis, we transformed the C_{biom} (g m⁻²) data for both sample types using the natural logarithm transformation ($C_{\text{biom}}' = \ln(C_{\text{biom}})$), in order to meet the normality assumption for ANOVA analyses. The average carbon content in vegetation and litter layer biomass was 87.65 and 80.29 g m⁻², respectively (Table 1-3-3).

The biomass carbon content results indicated that, despite significantly higher C_{biom} (wt%) concentration of the vegetation biomass for strata of FSQC II compared to FSQC I, the estimated carbon content per unit area, C_{biom} , g m⁻² was not significantly different. The observed significant differences in biomass C_{biom} (wt%) concentration could be due to difference of grass and legume species that dominate the strata of different site quality. The biomass of dominant grass and legume species on strata of FSQC I contain significantly lower amount of carbon compared to the dominant species on strata of FSQC II.

Table 1-3-3. Vegetation and litter layer biomass carbon contents measured on the Flint Gap project site. All samples were collected in summer 2005.

Type	Mean	Std. Error	Min	Max	N
	----- g C m ⁻² -----				
Vegetation	87.65	3.347	16.36	386.06	218
Litter layer	80.29	3.729	2.53	279.99	211

Despite the similar C_{biom} (g m⁻²) content across strata of different FSQC, because of the superior soil conditions on strata of FSQC I, the grass and legume plants could develop faster and grow less-dense plant biomass of lower carbon content. In contrast, due to the relatively inferior soil conditions on strata of FSQC II, the dominant plant species could develop slower and grow more-dense plant biomass of relatively higher carbon content.

Similar to the sampling intensity estimation for SOC measurement described above, we used the data in Table 1-3-3 to determine the number of vegetation and litter layer biomass samples required to measure C_{biom} (g m⁻²) content on Flint Gap at accuracy levels ranging from 10 to 25% and probability levels ranging from 80 to 99% (Figure 1-3-6).

If the 20%-accuracy and 90%-precision levels were the reporting standards for carbon credits in a future carbon market, 3 and 4.2 biomass samples ha⁻¹ for the vegetation and litter layer, respectively, would be sufficient to estimate the C_{biom} (g m⁻²) on the Flint Gap site (Figure 1-3-6). Similar to the SOC sampling intensities in Figure 1-3-4, significantly higher sampling intensities would be required to estimate biomass carbon content at higher levels of accuracy and precision (Figure 1-3-6). The sampling intensities for litter layer biomass were slightly higher than vegetation biomass due to the higher variation of litter layer biomass carbon compared to vegetation biomass (Table 1-3-3).

The sampling intensities for biomass carbon estimation on Flint Gap were one order of magnitude smaller compared to SOC (Figures 1-3-4 and 1-3-6). This was expected. During site reclamation, most mine sites are hydro-seeded to grass and legume species resulting in a relatively uniform distribution of seeds across the site. In contrast, within a mapping strata of uniform FSQC, there might be small areas (patches of land) that have very different chemical and physical properties from areas only a few meters apart.

Although there were no implications of site quality mapping on measuring the carbon content in vegetation biomass on mined lands sown to grass, mined land stratification was significantly important for the proper analysis of the SOC pool in these systems. Similarly, other studies have shown that the effects of site quality could be significant on the carbon sequestration potential of forests established on surface-mined land (Amichev et al. 2004). Therefore, it is logical that site quality mapping, and the resulting site stratification of a mined land to strata of homogenous FSQC, be accomplished at the start of any carbon sequestration project on mined land. In addition to making carbon accounting, verification, and reporting procedures easier, site quality data for a mined site could also be used for planning project activities and selecting suitable land management scenarios, so that mined land owners' objectives are met, while maximizing the carbon sequestration potential of mined land.

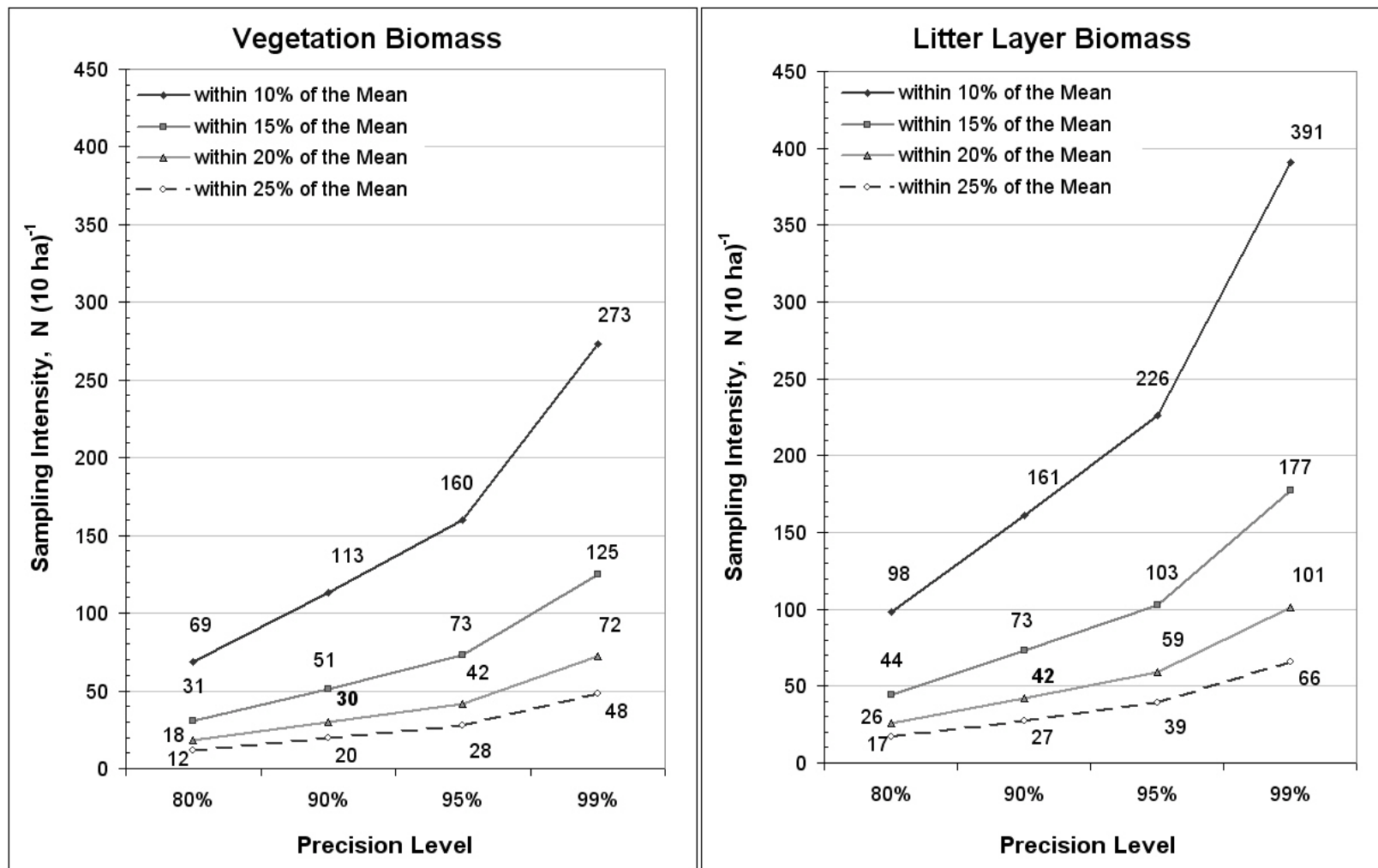


Figure 1-3-6. Sampling intensities for vegetation and litter layer biomass $C_{\text{biom}}(\text{g m}^{-2})$ stock determination at four accuracy (10, 15, 20, and 25% from the mean) and four precision levels (80, 90, 95, and 99%) on the Flint Gap mined site.

Because the sampling intensities for biomass carbon measurement were smaller than the SOC sampling intensity, some of the assigned permanent soil sampling locations in Figure 1-3-5 could also be used as permanent vegetation biomass sampling locations. For example, for the illustrated mapping strata in Figure 1-3-5, which had 0.88 ha area, 4 ($0.88 \times 3 + 0.88 \times 3 \times 20\%$) and 5 ($0.88 \times 4.2 + 0.88 \times 4.2 \times 20\%$), including 1 reserve plot each, should be selected at random and marked as permanent sampling plots for vegetation and litter layer biomass carbon content estimation, respectively, at the desired accuracy and precision levels.

Conclusions

Forest establishment on reclaimed mined lands is regarded as a viable method to sequester significant amounts of atmospheric CO₂ in the mine soil, live vegetation biomass, and harvested wood products. The successful establishment of trees on mined lands would also create environmental, economic, and social benefits for adjacent human communities. The main premise to use reforestation on mined lands is that these forests would sequester atmospheric CO₂ at rates greater in magnitude than the current rates under hayland/pasture land-use conditions.

The purpose of this study was to develop and evaluate a mined land-specific sampling protocol for ecosystem carbon analysis that would enable mined land owners to measure and report sequestered carbon in the form of vegetation and litter layer biomass, and SOC. Based on our experience and knowledge for mined site characterization and mapping, horizontal and vertical variability of SOC, and the results for minimum detectable difference of carbon stocks and maximum cost-effective depth of analysis in mine soils, we established the necessary procedures to compute accurate and precise ecosystem carbon content estimates for mined lands at various stages of floristic succession, ranging from abandoned grasslands to shrubland and forest land. Organized in a four-step approach, we developed a universal field measurement protocol for carbon inventory on mined lands, which could be applied to any mined site in the Appalachian coalfield. The four-step procedures aimed at: (step A) establishing permanent carbon monitoring plots across a mined site; (step B) baseline carbon stock estimation; (step C) periodic carbon content measurement; and (step D) reference areas.

The results in this substudy, which were used to evaluate the procedures in step A, proved practical and useful for a prototype carbon sequestration project established in Russell County, Virginia. The sampling intensities for SOC estimation at the project site at the 90% precision and 20% accuracy levels for strata of FSQC classes I, II, III, and IV were 27, 37, 39, and 40 soil sampling plots ha⁻¹, respectively. At similar accuracy and precision levels, 3 and 4.2 biomass samples ha⁻¹ were the sampling intensities for vegetation and litter layer carbon content, respectively. The proposed analysis methods could be used for sampling intensity estimation on other carbon projects on mined land so that the minimum accuracy and precision standards are met for a future carbon inventory market in the U.S.

Task 1, Sub-study 4: Carbon Sequestration in Forests on Mined Land

Experimental

Because the carbon sequestration potential of forested mined land is not well known, it must be characterized in order to make comparisons with other carbon sequestration projects, and to better understand the differences in carbon capture levels under varying forest and mined land conditions. Therefore, we characterized 14 mined and eight non-mined forests, adjacent to the mined forests, throughout the Midwestern and Appalachian coal regions to accomplish the goals of this project. The objectives of this project were to: (1) estimate and compare the ecosystem carbon content on disturbed and undisturbed forest land; (2) determine the effects of mine soil quality and stand age on the carbon sequestration potential of forested mined land; and (3) develop empirical models for carbon sequestration in tree biomass, litter layer, and soil of coniferous and deciduous forests established on mined land.

Carbon Data Set

The data for this project were obtained from a previous study conducted on 14 reforested mine sites located on mined land reclaimed prior to the passage of the Surface Mining Control and Reclamation Act (SMCRA) of 1977, and eight reference (non-mined), adjacent forests. The studies by Rodrigue (2001) and Rodrigue and Burger (2004) were designed using the classic retrospective research approach of evaluating long-term response of forests to treatments imposed at an earlier time (Powers et al. 1994). Retrospective studies look back in time to assess the effects of past treatments or conditions in the present. The principal advantage of retrospective studies is expediency in weighing the long-term effect of earlier events. The main disadvantage is the uncertainty associated with the precise conditions that existed when the treatments were imposed. Because the investigator usually has nothing to do with the application of the treatments, pre-treatment characterization data may not be available; pre-treatment conditions must be established from historical records. Nonetheless, retrospective studies provide an indispensable and powerful way of studying long-term phenomena such as forest growth and development (Burger and Powers 1991).

For this study, forest vegetation and litter layer plots were described, measured, and sampled by Rodrigue (2001). Soil pits were dug to approximately 1.5 m depth, and soil profiles were characterized and sampled at four randomly chosen plots within each forest stand. A more detailed description of the site selection criteria, study locations, and data collection approach can be obtained from Rodrigue (2001) and Rodrigue and Burger (2004). A summary of their procedures follows.

Fourteen pre-SMCRA forested sites across seven states, each with an average size of 2.5 ha of contiguous forest cover, were located on mined lands in the Midwestern and Appalachian coalfields of the U.S. (Figure 1-4-1). The location and forest site types were chosen to represent the mining and reclamation conditions that existed in the Midwestern and Appalachian coalfields prior to the passage of SMCRA (Rodrigue 2001). After visual examination of all potential locations, sites for measurement were chosen based on adequate stand size, stand age (indicating pre-SMCRA planting), forest conditions present, and the presence of an adjacent, suitable, and comparable non-mined forest (Rodrigue 2001).

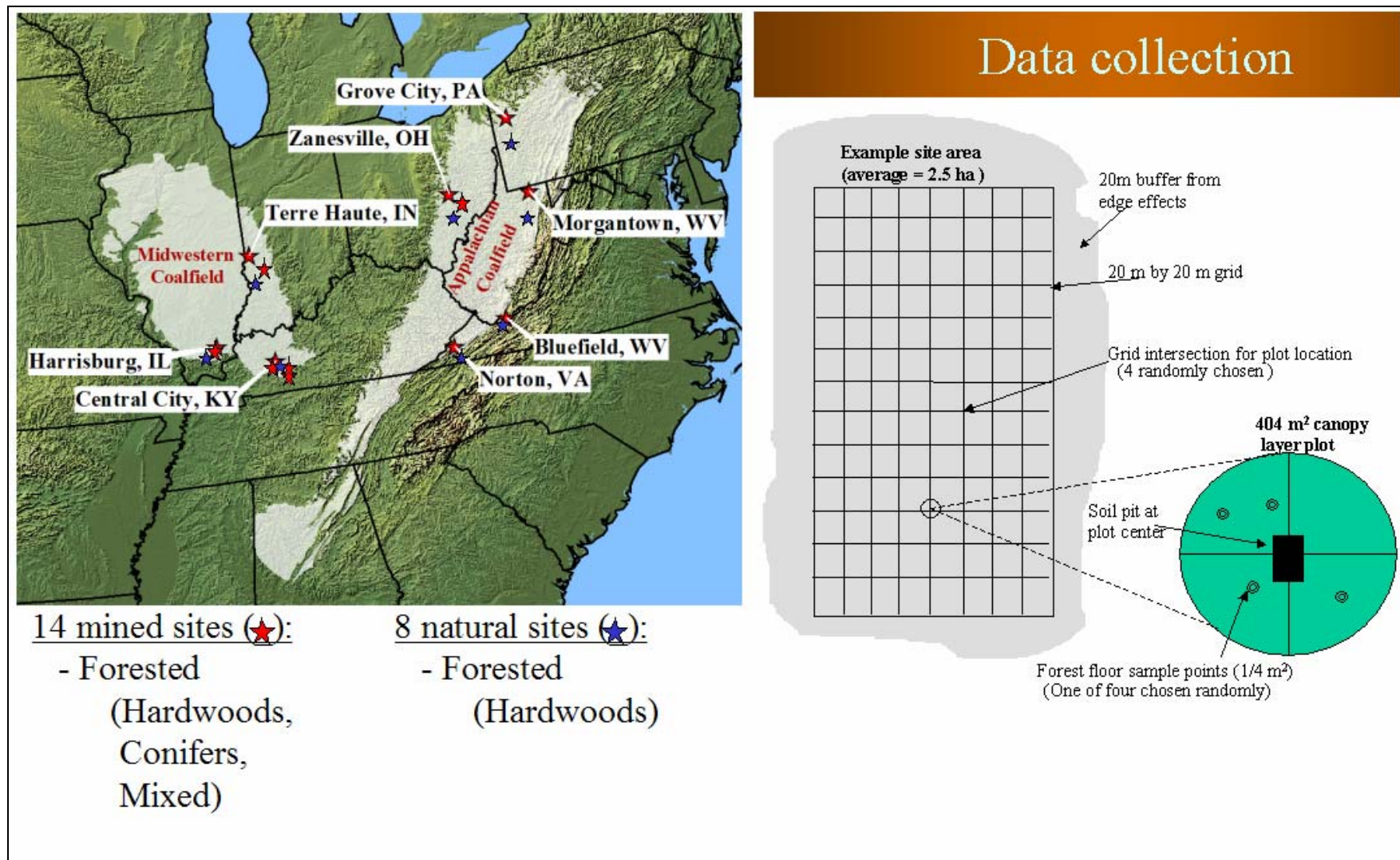


Figure 1-4-1. Mined and non-mined forested study site approximate location and data collection procedures used by Rodrigue (2001) and Rodrigue and Burger (2004).

Adjacent to the mined sites, reference native forests were located and measured. The non-mined forests were chosen based on the same criteria as the mined forests. With assistance from local experts and with the use of stand and soil maps, Rodrigue (2001) located forest stands on natural, undisturbed soils that best represented the local forest and soil types. The selected non-mined forests represented the topography, soil, and forest conditions similar to those present on the mine sites before they were disturbed. All non-mined sites were mature, well-stocked, native forest stands. Natural soils included Inceptisols, Ultisols, and Alfisols (Rodrigue 2001; Rodrigue and Burger 2004).

A 20 x 20-m grid was superimposed across the sites after the boundaries were established. A 20-m buffer strip was maintained on all edges of each forest site (Figure 1-4-1). Grid lines were placed perpendicular to the banks of open-pit mined sites where more than one spoil bank existed to ensure that the site's microtopography was taken into account. All subsequent sampling was based from intersections of the grid.

Measurement plots were established at four randomly chosen grid intersections within the selected mined and non-mined forests, so that the grid intersections became the centers of the individual measurement plots. Tree, litter, and soil data were collected from each forest plot independently from all other plots. Field data collection took place between May and August 1999, with the exception of two sites, which were measured in August 1998.

Tree cores were used to determine stand age of the dominant and co-dominant trees. The age of the mined forest plots ranged from 18 to 71 years old, with an average age of 39 years (Table 1-4-1). The canopy layer species ranged from pure hardwood and pine stands to mixed stands. All mined sites were planted with trees produced in local nurseries. Mixed hardwoods dominated the non-mined, natural stands. The age of the non-mined forest plots ranged from 36 to 87 years old, with an average age of 55 years (Table 1-4-1).

Rodrigue (2001) measured tree diameter, height and litter layer biomass on each plot using the data collection scheme shown in Figure 1-4-1. Trees in the main canopy greater than 13.0 cm in diameter at breast height (DBH) were tallied within a 404-m² circular plot. Litter layer biomass estimates were determined from four 0.25-m² random samples at each plot and bulked to form a 1-m² sample. Bulk samples were dried, ground, and total carbon was determined with a LECO carbon analyzer. Litter layer estimates were corrected for ash and the mineral material that the samples contained.

At the center of each plot, Rodrigue (2001) dug a soil pit to a depth of 1.5 m (Figure 1-4-1) to characterize mine soil development and to measure the SOC content. In the field, pits were described using standard soil survey techniques to obtain total depth, horizon depth, and percent coarse rock fragments greater than 7.6 cm. Loose samples and duplicate bulk density samples were collected from each horizon (Rodrigue, 2001). In the laboratory, soil samples were air-dried, sieved (2 mm), and weighed to determine coarse rock fragments (≤ 7.6 cm). All soil carbon determination procedures were performed on the sieved 2-mm fraction. Soil properties were analyzed on samples from all horizons found in the profiles. Bulk density was determined using soil cores and was corrected for coarse rock fragments. Soils on the mined sites were classified as Udorthents (Rodrigue 2001; Sencindiver and Ammons 2000; Zeleznik and Skousen 1996).

Table 1-4-1. General description of 55 mined and 32 non-mined (adjacent) forest plots in the Midwestern and Appalachian coalfields, including dominant canopy species, stand age, and site index (height of white oak at 50 years of age). Basal area estimates were computed for trees with DBH greater than 13 cm. More details can be obtained from Rodrigue (2001).

State	Site	Stand Age †	Site Index	Canopy Type	Tree Basal Area		Total Tree Basal Area
		years	meters		hardwoods	pinus ‡	
MINED LAND							
Illinois	IL 2 - G4	41	23.6	Cottonwood	100	0	16,164
	IL 2 - E1	44	24.1	Cottonwood	100	0	16,990
	IL 2 - C2	45	24.1	Cottonwood	100	0	10,570
	IL 2 - A2	46	24.8	Cottonwood	100	0	12,561
	IL 1 - E2	47	28.5	White oak / tulip poplar	100	0	12,480
	IL 1 - I1	47	27.1	White oak / tulip poplar	100	0	11,690
	IL 1 - B1	49	29.9	White oak / tulip poplar	100	0	10,768
	IL 1 - G2	53	26.2	White oak / tulip poplar	100	0	16,235
	Average	47	26.0		100	0	13,432
Indiana	IN 2 - D3	35	27.1	Pitch pine	25	75	5,249
	IN 2 - F10	43	27.1	Pitch pine	59	41	7,499
	IN 2 - A9	44	24.1	Pitch pine	38	62	10,185
	IN 2 - E6	44	18.3	Pitch pine	86	14	5,928
	IN 1 - E3	46	21.5	Mixed hardwoods / Pitch pine	25	75	10,666
	IN 1 - G2	46	21.5	Mixed hardwoods / Pitch pine	53	47	11,320
	IN 1 - B7	47	24.5	Mixed hardwoods / Pitch pine	19	81	15,951
	IN 1 - D1	54	25.9	Mixed hardwoods / Pitch pine	23	77	6,572
	Average	45	23.8		41	59	9,171
Kentucky	KY 4 - L1	30	21.6	Loblolly pine	4	96	17,299
	KY 1 - B4	31	22.7	Mixed hardwoods	100	0	9,602
	KY 1 - G1	31	24.1	Mixed hardwoods	100	0	14,051
	KY 1 - F2	32	21.6	Mixed hardwoods	100	0	7,155
	KY 1 - H2	32	21.8	Mixed hardwoods	100	0	9,334
	KY 1 - J2	32	23.2	Mixed hardwoods	100	0	7,793
	KY 3 - A4	34	27.1	E. white pine / loblolly pine	26	74	13,205
	KY 4 - M3	34	22.6	Loblolly pine	10	90	25,685
	KY 4 - X2	35	23.8	Loblolly pine	5	95	16,873
	KY 2 - A5	36	25.8	Mixed hardwoods	100	0	7,692
	KY 2 - G5	37	25.2	Mixed hardwoods	100	0	11,026
	KY 3 - B2	38	22.1	E. white pine / loblolly pine	18	82	12,328
	KY 3 - D1	39	25.1	E. white pine / loblolly pine	31	69	11,193
	KY 2 - E4	40	23.5	Mixed hardwoods	100	0	12,399
	KY 3 - C3	40	25.5	E. white pine / loblolly pine	26	74	17,400
	KY 2 - K1	41	27.0	Mixed hardwoods	100	0	9,921
	Average	35	23.9		64	36	12,685
Ohio	OH 2 - B2	35	25.9	Mixed hardwoods	100	0	11,989
	OH 1 - A1	42	23.9	Mixed hardwoods	100	0	11,543
	OH 2 - D4	46	25.6	Mixed hardwoods	100	0	11,320
	OH 1 - D4	47	25.0	Mixed hardwoods	100	0	10,504
	OH 1 - B3	50	25.3	Mixed hardwoods	100	0	13,605
	OH 2 - C3	51	27.7	Mixed hardwoods	100	0	13,012
	OH 1 - F5	54	27.1	Mixed hardwoods	100	0	10,920
	OH 2 - B7	71	28.7	Mixed hardwoods	100	0	15,647
	Average	50	26.2		100	0	12,317

Table 1-4-1 (continued).

State	Site	Stand Age †	Site Index	Canopy Type	Tree Basal Area		Total Tree Basal Area
		years	meters		hardwoods	pinus ‡	
					---- % of total ----	cm ²	
Pennsylvania	PA 1 - C5	36	19.2	E. white pine / Scotch pine	6	94	11,335
	PA 1 - F2	38	21.0	E. white pine / Scotch pine	5	95	12,703
	PA 1 - A4	41	20.7	E. white pine / Scotch pine	13	87	16,863
	PA 1 - B6	42	22.6	E. white pine / Scotch pine	2	98	9,369
	Average	39	20.9		7	93	12,568
Virginia	VA 1 - P2	18	26.2	E. white pine	5	95	8,974
	VA 1 - P3	19	25.9	E. white pine	0	100	7,084
	VA 1 - P1	20	23.2	E. white pine	0	100	4,474
	Average	19	25.1		2	98	6,844
West Virginia (north)	WV 1 - F3	31	17.4	E. white pine	31	69	10,681
	WV 1 - D1	32	18.3	E. white pine	34	66	13,347
	WV 1 - B3	38	16.2	E. white pine	30	70	13,554
	WV 1 - E3	38	14.9	E. white pine	61	39	9,602
	Average	35	16.7		39	61	11,796
(south)	WV 2 - A2	27	22.9	E. white pine	24	76	17,091
	WV 2 - C4	27	32.0	E. white pine	3	97	25,999
	WV 2 - G2	27	29.9	E. white pine	18	82	18,642
	WV 2 - D2	31	25.9	E. white pine	5	95	15,718
	Average	28	27.7		12	88	19,363
<u>NON-MINED LAND (adjacent to mined sites)</u>							
Illinois	IL - A2	39	33.8	Scarlet oak / red maple	100	0	14,806
	IL - G4	39	25.9	Scarlet oak / red maple	100	0	14,730
	IL - F8	43	24.4	Scarlet oak / red maple	100	0	12,749
	IL - B9	52	27.4	Scarlet oak / red maple	100	0	12,576
	Average	43	27.9		100	0	13,715
Indiana	IN - B1	36	24.4	Oak / tulip poplar	100	0	9,506
	IN - C4	41	25.3	Oak / tulip poplar	100	0	10,408
	IN - D6	50	20.4	Oak / tulip poplar	100	0	9,374
	IN - D1	62	28.7	Oak / tulip poplar	100	0	18,424
	Average	47	24.7		100	0	11,928
Kentucky	KY - J	40	24.1	Oak / tulip poplar	100	0	7,636
	KY - O	48	22.4	Oak / tulip poplar	100	0	9,805
	KY - C	50	25.0	Oak / tulip poplar	100	0	14,441
	KY - E	69	23.5	Oak / tulip poplar	99	0	8,802
	Average	52	23.7		100	0	10,171
Ohio	OH - C10	51	22.9	Oak / tulip poplar	100	0	9,653
	OH - C4	55	21.0	Oak / tulip poplar	100	0	9,024
	OH - A3	56	29.0	Oak / tulip poplar	100	0	9,212
	OH - E3	72	20.4	Oak / tulip poplar	100	0	9,237
	Average	58	23.3		100	0	9,282
Pennsylvania	PA - G2	54	26.5	Oak / tulip poplar / cherry	100	0	10,499
	PA - C2	57	23.5	Oak / tulip poplar / cherry	100	0	3,846
	PA - A2	64	27.4	Oak / tulip poplar / cherry	100	0	23,319
	PA - J1	64	22.9	Oak / tulip poplar / cherry	100	0	9,379
	Average	60	25.1		100	0	11,761

Table 1-4-1 (continued).

State	Site	Stand Age †	Site Index	Canopy Type	Tree Basal Area		Total Tree Basal Area
		years	meters		hardwoods	pinus ‡	
					---- % of total ----		cm ²
Virginia	VA - B6	62	27.1	Oak / tulip poplar	100	0	17,370
	VA - E1	63	27.7	Oak / tulip poplar	100	0	13,747
	VA - C7	71	25.9	Oak / tulip poplar	100	0	10,610
	VA - A2	87	28.3	Oak / tulip poplar	100	0	15,049
	<i>Average</i>	<i>71</i>	<i>27.3</i>		<i>100</i>	<i>0</i>	<i>14,194</i>
West Virginia (north)	WV 1 - A2	55	30.8	Oak / tulip poplar	100	0	13,022
	WV 1 - E4	62	25.6	Oak / tulip poplar	100	0	14,137
	WV 1 - D4	65	21.2	Oak / tulip poplar	100	0	14,841
	WV 1 - B4	67	24.4	Oak / tulip poplar	100	0	10,423
	<i>Average</i>	<i>62</i>	<i>25.5</i>		<i>100</i>	<i>0</i>	<i>13,106</i>
(south)	WV 2 - B1	38	28.0	Oak / tulip poplar	100	0	7,763
	WV 2 - D5	49	24.2	Oak / tulip poplar	100	0	16,326
	WV 2 - A6	54	25.9	Oak / tulip poplar	100	0	11,381
	WV 2 - C2	62	25.6	Oak / tulip poplar	100	0	13,696
	<i>Average</i>	<i>51</i>	<i>25.9</i>		<i>100</i>	<i>0</i>	<i>12,291</i>

† Represents average age of trees measured on sampling plots.

‡ Eastern white pine (*Pinus strobus*), loblolly pine (*Pinus taeda*), pitch pine (*Pinus rigida*), Scotch pine (*Pinus sylvestris*), Virginia pine (*Pinus virginiana*).

Rodrigue (2001) and Rodrigue and Burger (2004) determined soil organic carbon content using the Walkley-Black wet oxidation procedure (Nelson and Sommers, 1982) assuming that soil coal fragments, present in most mine soils, would not interfere with the SOC estimation. However, in order to account for the effect of coal fragments on SOC estimates in the soils analyzed, we corrected the Walkley-Black carbon values reported by Rodrigue (2001) and Rodrigue and Burger (2004) using the coal-correction equation described in Substudy 1. The linear regression equation ($R^2=55\%$) used to determine the weight percent soil organic carbon (SOC, wt%) from measured Walkley-Black carbon values (WB-SOC, %) for mine soils developed from sandstone (SS) siltstone (SiS) or approximately 50:50% mixture of SS and SiS mine spoil materials was as follows:

$$SOC(wt\%) = 0.11063 * (Order) + 0.28687 * (WB - SOC, \%)$$

where SOC (wt%) = true soil organic carbon content expressed as percent by sample weight; Order = 1 for sandstone, = 2 for siltstone, and = 3 for 50:50 SS:SiS material; WB-SOC, % = Walkley-Black soil organic carbon measurement result. The corrected soil carbon estimates, SOC (wt%) were converted to tons per ha ($Mg\ C\ ha^{-1}$) using the bulk density and volume of the fines, and layer depth data for each soil pit in the data set from Rodrigue (2001).

Geostatistical Analysis

Before performing any statistical analyses, we determined the auto-correlation of the data (the dependence with respect to separation distance between individual plots) using the semivariance statistic $\gamma(h)$ computed for a range of distance intervals h (Burgess and Webster 1980; Ettema and Wardle 2002; Robertson 1987) as follows:

$$\gamma(h) = \frac{1}{2N(h)} \sum_{i=1}^n [z(s_i) - z(s_i + h)]^2$$

where $N(h)$ = number of observations (i.e., plots) separated by distance h ; $z(s_i)$ = value of a certain ecosystem variable at location s_i ; and $z(s_i + h)$ = value at a location that is at distance h away from s_i . The results of the geostatistical analysis were used to determine whether the plot data points were independent of each other and suitable for statistical analysis. We used the PROC VARIOGRAM procedure in SAS® statistical software to perform all geostatistical analyses of the data.

Calculations and Comparison of the Ecosystem Carbon Sequestration on Mined and Non-Mined Forest Land

For all measurement plots, we estimated above-ground tree biomass and coarse-root biomass (kg dry wt) from species-specific tree diameter data for each tree tallied by Rodrigue (2001). These estimates were based on regression equations described in Jenkins et al. (2003). The above-ground and coarse-root weights were summed to produce an estimate of the total tree biomass in dry weight units. Total tree biomass included stem wood, stem bark, foliage, treetops, branches, stumps, and coarse roots. Total tree biomass was converted to kilograms of carbon with a conversion factor for different regional species groups (Birdsey 1992). Total tree carbon, litter layer carbon, and soil organic carbon in kilograms per hectare were converted to metric tons per hectare (Mg C ha^{-1}). The ecosystem carbon content was estimated as the sum of the total tree, litter layer, and soil carbon pools.

Carbon in ground-layer woody or herbaceous biomass was not included in our analysis because carbon estimates could not be generated for this portion of the forest ecosystem. However, carbon contained in these understory components is often ignored in biomass estimates because it only amounts to 1% to 2% of the above-ground carbon content (Birdsey 1992).

Effects of Mine Site Quality and Stand Age on the Carbon Sequestration Potential of Forested Mined Land

Foresters often use site indices (height of co-dominant trees at a selected age; e.g., age 50) to indicate the productive potential of a forest site. To produce comparable site quality estimates for the mined and non-mined measurement plots, the site index of each measured tree species on all plots was standardized to the site index (SI) of a single species, white oak at base age 50 years (Doolittle 1958). Site index data for each forested plot were obtained from Rodrigue (2001), who converted the site index of measured tree species to the site index of white oak using site index tables and graphs that were most suitable for each plot's location (Carmean et al. 1989).

We queried the carbon data from Rodrigue (2001) to create separate data sets for pine, mixed, and hardwood stands developed on mined land. Carbon content in vegetation biomass, litter layer, soils, as well as stand age, and site index data were extracted from the 55 mined land measurement plots for separate regression analyses regressing carbon content with stand age and site index in SAS® statistical software (SAS 2007). Because many of the mined plots in this study were comprised of both hardwood and pine species (Table 1-4-1), we defined each plot as follows: pine stands were forest stands in which the basal area of all hardwood trees (BA_{HW})

was lower than or equal to 20% of total stand basal area, i.e., $BA_{HW} \leq 20\%$; mixed stands were forest stands with BA_{HW} ranging from 20 to 80%; and hardwood stands were forest stands with $BA_{HW} > 80\%$.

Using the plot BA estimates as a criterion for tree stocking, we determined whether any plots from the mined pine stand data set were poorly stocked, and whether the reasons for the observed stocking anomalies could be justified. Similarly, tree-stocking data for the mined mixed and hardwood stand data sets and for the non-mined hardwood stands were used to further evaluate the respective measurement plots. Poorly stocked plots that were confirmed as data outliers were removed from the respective data sets before performing any statistical analyses.

For the three mined forest types, we performed correlation analyses in SAS® to determine the relationship between SI, Age, and SI/Age, as independent variables, and tree, litter layer, and soil carbon content, as the dependent variables. We also correlated SI, Age, and SI/Age with the total ecosystem carbon content to determine the relationship between stand and site conditions and the carbon sequestration potential of the mined sites. Standard statistical tools, the PROC CORR procedure, were used in SAS® to perform these analyses.

We performed multiple regression analyses of ecosystem carbon content by forest type measured on mined land to determine the effects of site quality and stand age on the carbon sequestration potential of mined land under various forest cover. We used standard regression procedures (PROC REG and the Mallows' Cp model selection option) in SAS® to generate the regression equations using SI and Age as independent variables, and ecosystem carbon as the dependent variable. We based our multiple regression analyses on well-established and widely-used tree growth and yield models (Brender 1960; Brender and Clutter 1970; Clutter 1963; Nelson et al. 1961). The general form of the regression model was:

$$\ln(C_{Ecosystem}) = b_0 + b_1(SI) + b_2(Age) + b_3(SI^2) + b_4\left(\frac{1}{Age}\right) + b_5\left(\frac{1}{Age^2}\right) + b_6\left(\frac{SI}{Age}\right)$$

where $\ln(C_{Ecosystem})$ = natural log of ecosystem carbon content ($Mg\ C\ ha^{-1}$); b_i = regression coefficient, $i=0,1,2,\dots,6$; SI = site index (tree height in meters for white oak at base age 50 years); Age = stand age.

The resulting empirical models were plotted on a 3-dimensional (3D) diagram to depict the distribution of ecosystem carbon sequestered by mixed, pine, and hardwood stands across the spectrum of SI and stand age on mined land. The 3D representation of ecosystem carbon estimates as a function of SI and age allowed for better visualization of the combined effects of both parameters on the potential of mined sites to sequester carbon.

Projecting Carbon Data to Rotation Age 60 Years

To achieve our objective of adequately comparing carbon sequestration in forests on mined and non-mined land, and among forests of different forest types on mined land, we projected the current carbon pools in all mined and non-mined plots to the carbon pools expected at rotation age of 60 years. We used the tree age projection techniques described by Rodrigue (2001) “to increase” (e.g., mined plot IL 2–G4, and non-mined plot IL–A2) and “to reduce” (e.g., mined: IN 2–A9, and non-mined: IN–D1) the present tree biomass of the mined and non-mined forest stands to the biomass levels expected at rotation age (60 years). Foresters define rotation age as

the number of years between the time a stand regenerates, or is planted, and the time when the mature trees are harvested for timber (Nyland 1996).

Rodrigue (2001) developed regression equations for predicting the individual tree DBH increment in a certain forest at any stand age by using tree measurements of the stem core for the last 10 years of stand development and bark thickness by tree species groups. The following was the general form of the linear regression model used by Rodrigue (2001):

$$\ln(Core_{10}) = b_0 + b_1 * \left(\frac{1}{DBH_{10}} \right)$$

where $\ln(Core_{10})$ = natural log of the 10-year DBH increment; b_0 = intercept coefficient; b_1 = slope coefficient; DBH_{10} = tree diameter outside bark at 10 years prior to current age. Species with low occurrences on the plots that had similar growth characteristics were grouped together in order to increase the amount of tree measurement data used in the regression analysis per tree species group (Rodrigue, 2001). Examples of the species analyzed include *Pinus rigida* P. Mill. (pitch pine), *Pinus taeda* L. (loblolly pine), *Pinus sylvestris* L. (Scots pine), *Robinia pseudoacacia* L. (black locust), *Prunus serotina* Ehrh. (black cherry), *Quercus alba* L. (white oak), *Quercus rubra* L. (northern red oak), *Nyssa sylvatica* Marsh. (blackgum), *Platanus occidentalis* L. (American sycamore), *Liquidambar styraciflua* L. (sweetgum), and *Ulmus spp.* (elm). Examples of species groups analyzed include poplars (*Liriodendron tulipifera* L. (tulip poplar), *Magnolia acuminata* L. (cucumbertree), *Magnolia fraseri* Walt. (Frasier magnolia)), maples (*Acer rubrum* L. (red maple), *Acer saccharinum* L. (silver maple), *A. saccharum* Marsh. (sugar maple), *A. negundo* L. (boxelder)), cottonwood (*Populus deltoides* Bartr. ex Marsh. (eastern cottonwood), *P. tremuloides* Michx. (aspen)), ash (*Fraxinus americana* L. (white ash), *F. pennsylvanica* Marsh. (green ash)), other oaks (*Quercus coccinea* Muenchh. (scarlet oak), *Q. velutina* Lam. (black oak), *Q. prinus* L. (chestnut oak), *Q. imbricaria* Michx. (shingle oak)), birch (*Betula lenta* L. (black birch), *B. nigra* L. (river birch), *B. alleghaniensis* Britt. yellow birch)), and hickory (*Carya glabra* P. Mill. (pignut hickory), *C. ovata* P. Mill. (shagbark hickory), *Juglans nigra* L. (black walnut)) (Rodrigue 2001).

Using the species-specific coefficients b_0 and b_1 in Rodrigue (2001), we estimated tree size-specific, 1-year DBH increment (cm yr^{-1}) for all individual trees in each mined and non-mined plot. We extracted individual tree DBH data from each forest plot using data query procedures in Microsoft® Access database management software. To determine the expected DBH change of individual trees during the period of years prior, or past, rotation age, we multiplied the tree size-specific, 1-year DBH increment by the number of years before or after rotation age. Finally, we added the expected DBH change to the current DBH measurement to compute the DBH at rotation age.

For example, using the regression equation for pitch pine, $\ln(Core_{10}) = -0.2856 + 0.9579 (1/DBH_{10})$ (Rodrigue 2001), we estimated that the tree size-specific, 1-year DBH increment for a 10.2-cm (~ 4 in) and a 63.5-cm (~25 in) tree would be 0.24 cm (0.095 in, estimated as $\exp(-0.2856 + 0.9579*(1/4)/10)$) and 0.20 cm (0.078 in, estimated as $\exp(-0.2856 + 0.9579*(1/25)/10)$), respectively. Assuming that both pitch pine trees were measured in a forest plot of stand age 45 years, we estimated that the DBH of the smaller-diameter tree at rotation age 30 years (i.e., 15 years ago) was 6.6 cm (estimated as $10.2 - 15*0.24$) and was 60.5 cm for the larger-diameter tree. We used a similar approach to increase the DBH of individual trees in forest stands of age less than the rotation age. We estimated the total projected, plot-level total tree

biomass as the sum of individual tree biomass predictions using the projected, rotation-age DBH estimates. We projected pine and mixed plots to rotation age 30 years, and projected the mined and non-mined hardwood plots to age 60 years. We estimated above-ground tree biomass and coarse-root biomass using the regression equations as a function of DBH described in Jenkins et al. (2003).

In order to estimate the total amount of sequestered carbon in tree biomass and harvested wood after two successive, 30-year rotations for pine and mixed stands, we assumed that 24% of the carbon in harvestable tree biomass from the first rotation would remain sequestered in wood products in use and in landfills 30 years after the harvest (derived from Smith et al. 2006). The harvestable tree biomass included stem wood, defined as tree stem wood from 30.48 cm stump height to 10.16 cm top diameter outside bark (Jenkins et al. 2003), stem bark, foliage, treetops, and branches. We also assumed that stand management practices would remain the same during the second 30-year rotation period ensuring similar tree growth and forest productivity within these stands. Therefore, we estimated the expected total amount of sequestered carbon in vegetation biomass (standing trees and wood products) in pine and mixed stands at the end of two consecutive, 30-year rotations by adding the total tree carbon content from the second rotation, including stem wood, stem bark, foliage, treetops, branches, stumps, and coarse roots, to the carbon amount stored in wood products in-use and in-landfills manufactured from harvested tree biomass from the first rotation (estimated as 24 % of the carbon in harvestable tree biomass, including stem wood, stem bark, foliage, treetops, and branches).

Forest floor biomass accumulation is a function of many environmental factors as well as tree species composition (Fisher and Binkley 2000; Smith and Heath 2002). Considering that the natural stands measured by Rodrigue (2001) grew under favorable climatic conditions characteristic of a typical temperate region of the United States, and the fact that these natural stands, on average, were 55 years of age, and comprised primarily of mixed hardwood species, we assumed that the litter layer biomass and, hence, the litter layer carbon pool was in equilibrium (Fisher and Binkley 2000; Smith and Heath 2002). Also, due to a paucity of consistent information in the literature relative to changes in soil carbon pools with time under tree vegetation cover on natural, non-mined sites, we assumed that the soil carbon pool in the natural sites had reached approximately steady state levels (Houghton and Hackler 2000). Therefore, we assumed negligible changes in soil carbon and litter layer carbon on natural sites during the stand projection period. On average, we projected the ecosystem carbon content 5 years into the future, or the past, in order to predict rotation age carbon levels, at age 60 years.

For the mined plots, we assumed that soil and litter layer carbon pools were not in equilibrium and that carbon was being sequestered in the soil and litter layer biomass at rates similar to afforested lands. We assumed that carbon sequestration in the soil carbon pool occurred at a constant rate until a new equilibrium was reached, similar to the assumptions made by Heath et al. (2002) in a similar carbon modeling study on afforested land. We assumed that the soil carbon accumulation rate ($\text{Mg C ha}^{-1} \text{ year}^{-1}$) could be approximated by dividing the present soil carbon content (Mg C ha^{-1}) by the stand age (years) in each mined plot. We assumed that the equilibrium state of soil carbon on forested mined sites could be approximated by the soil carbon levels measured on the adjacent, non-mined forest stands. For pine and mixed forest stands, we assumed that carbon would continue to accumulate in the soil carbon pool without significant deviations from the assumed rates following timber harvest at age 30 and immediate tree planting on the harvested sites (Griffiths and Swanson 2001; Johnson 1992; Johnson and

Curtis 2001). Therefore, we predicted the amount of carbon sequestered in the soil during a 60-year period (one 60-year and two 30-year rotations for hardwood, and for pine and mixed stands, respectively) by either multiplying the plot-specific soil carbon accumulation rate by 60, or by assigning the equilibrium soil carbon content measured on adjacent non-mined land, whichever was smaller.

For all mined plots, we assumed that the amount of carbon in the litter layer would increase asymptotically until a certain steady state level is reached, and that this accumulation could be modeled using the relationship between litter layer carbon and stand age described in Smith and Heath (2002). Smith and Heath (2002) modeled net accumulation of litter layer carbon as a function of age by forest type and region within the United States as follows:

$$\text{Litter layer C (Mg ha}^{-1}\text{) accumulation} = \frac{A * age}{(B + age)} \quad \text{Eq. 1}$$

where A = coefficient representing the upper limit of litter layer carbon mass within a certain mature forest ecosystem; age = stand age; B = coefficient estimated by assuming that the model passes through an ecosystem-specific data point (or set of data) for which both litter layer carbon (Mg ha⁻¹) and stand age are derived experimentally. For example, for a given ecosystem, if the upper limit of litter layer carbon was represented by coefficient A', and the ecosystem-specific data point was (L', age'), where L' is the litter layer carbon mass (Mg ha⁻¹) measured at stand age age', then B could be computed as (A*age'/L') - age'.

We adopted the litter layer carbon mass decay equation developed by Smith and Heath (2002) to predict the amount of residual litter layer carbon in pine and mixed stands following harvesting disturbance at age 30 years. We estimated the amount of litter layer carbon that would remain on the site at the end of the second 30-year rotation in pine and mixed stands using the following equation (Smith and Heath 2002):

$$\text{Residual litter layer C (Mg ha}^{-1}\text{) at age 60} = C * \exp\left(-\frac{30}{D}\right) \quad \text{Eq. 2}$$

where C = litter layer carbon mass accumulated under pine or mixed forest vegetation at the end of the first 30-year rotation (estimated by substituting 30 for age in Eq. 1); exp = exponential function; 30 = years after harvesting; D = litter layer mean residence time (for the North region of the U.S.) reported by Smith and Heath (2002, their Table 4).

To develop litter layer carbon accumulation equations for mined lands, we assigned mined-land specific coefficients A and B using the mined land data in this study. For each forest type (pine, hardwood, mixed), we estimated the maximum litter layer carbon mass for mature stands. Although defining when certain forest stands become mature forests is difficult (Smith and Heath 2002), we used it to identify the most likely point in time when a steady state of litter layer carbon mass is reached. We defined hardwood, pine, and mixed stands as mature when they reach ages 70, 40, and 40 years, respectively. We estimated coefficients A and B as follows:

$$A_i = \text{Maximum}_{i=1, j=1}^{3, N} \left(\frac{\text{LitterC}_{ij}}{\text{Age}_{ij}} * \text{Mature\_years}_i \right) \quad \text{Eq. 3}$$

$$B_{ij} = \left(\frac{A_i * Age_{ij}}{LitterC_{ij}} - Age_{ij} \right) \quad \text{Eq. 4}$$

where i (1 through 3) = forest type (pine, hardwood, mixed); j (1 through N) = number plots of forest type i ; A_i = litter layer accumulation coefficient indicating the maximum litter layer mass in mature forests on mined land of forest type i ; B_{ij} = litter layer accumulation coefficient for plot j of forest type i ; $LitterC_{ij}$ = litter layer carbon mass (Mg ha^{-1}) measured on plot j of forest type i ; Age_{ij} = stand age measured on plot j of forest type i ; $Mature\_years_i$ = stand age when the trees in plot ij of forest type i mature.

For hardwood stands, we estimated the amount of carbon in the litter layer at the end of a 60-year rotation by substituting 60, A_i (estimated in Eq.3), and B_{ij} (estimated in Eq.4) for the age , A , and B parameters, respectively, in Eq. 1. For pine and mixed forest stands, we estimated the amount of carbon in the litter layer at the end of two successive 30-year rotations by summing the carbon in the accumulating litter layer biomass, as a result of litter fall from regrowth during the second 30-year rotation, and the carbon in the residual litter layer biomass, following harvesting at age 30 years. We estimated the carbon in the accumulating litter layer biomass by substituting 30, A_i (estimated in Eq. 3), and B_{ij} (estimated in Eq.4) for the age , A , and B parameters, respectively, in Eq. 1. We estimated the carbon in the residual litter layer using Eq. 2 and the respective A_i (estimated in Eq. 3), and B_{ij} (estimated in Eq. 4) parameters for pine and mixed stands; we used 8.4 years as the mean residence time (parameter D in Eq. 2) for litter layer under both pine and mixed forest vegetation (Smith and Heath 2002, their Table 4).

Empirical Models for Ecosystem Carbon Sequestration at Rotation Age for Coniferous and Deciduous Forests on Mined Land

For a 60-year-planned carbon sequestration project, we assumed that there would be two timber harvesting events for pine and mixed stands, at ages 30 and 60 years, and one harvest for hardwood stands, at age 60 years. For this modeling work, we used the projected, rotation age carbon data sets for total tree, litter layer, and soil carbon pools for each forest type, the computational methods for which were described in the previous section.

We developed a set of linear regression models using standard regression procedures (PROC REG, Mallows' Cp model selection option) in SAS® in which the independent variable was site index, and the dependent variable was (i) 60-year projected ecosystem carbon; (ii) 60-year projected total tree carbon; (iii) 60-year projected litter layer carbon; and (iv) 60-year projected soil organic carbon. We developed these linear regression models for mined pine, mixed, and hardwood stands.

We also developed carbon sequestration models for the non-mined sites using similar methods of analysis, where site index was the independent variable and the non-mined 60-year projected carbon pools, ecosystem, total tree carbon, litter layer carbon, soil, were the dependent variables. We superimposed the non-mined hardwood regression models with those for the mined hardwood forests to better understand the differences and similarities between the two forest ecosystems. In particular, we were interested in whether the carbon sequestration potential of mined land was affected by coal mining practices.

Results and Discussion

Geostatistical analysis

The carbon data set by Rodrigue (2001) was originally compiled from measurement plots (four plots per forest stand) located within 14 mined and eight non-mined forests. We performed geostatistical analyses by forest type for each ecosystem parameter (SI and stand age) and carbon pool (ecosystem, total tree biomass, litter, and the soil) to determine the degree of independence among the plot data in all stands. If the data were independent at the distances at which they were collected, then each plot's data would be considered a separate data point to use in standard statistical analyses, as opposed to using a limited number of stand-level, average carbon estimates.

Geostatistical analysis is the preferred method of analysis for spatial data that can be auto-correlated; i.e., dependent with respect to distance and time (Robertson 1987). In ecological studies, Robertson (1987) described the potential problems associated with spatial data, where standard statistics cannot be used due to data dependence; i.e., data collected from sites that are located close in space (or time) tend to be more similar than at larger distances. When certain data are spatially dependent at a chosen scale of analysis, the semivariance is low at short distances between measurement locations (i.e., data points are correlated to each other), and increases for larger distances, often reaching an asymptote (called the *sill*) at a separation distance, referred to as the *range*, beyond which the data are spatially independent (Burgess and Webster, 1980; Robertson, 1987) (Figure 1-4-2). To describe the auto-correlation of spatial data, researchers often model the $\gamma(h)$ as a function of the distance interval h using theoretical semivariogram models, such as the spherical, the gaussian, the exponential, and the power semivariogram model (SAS 2007).

Theoretically, the intercept of the semivariogram model should be zero at zero separation distances. However, in reality, the intercept of semivariogram models often is greater than zero and is referred to as the nugget variance (Figure 1-4-2) representing the variance due to a combination of sampling error and spatial dependence not accounted for at the distances sampled. The ratio (*sill-nugget*)/*sill*, ranging from 0 to 1, is often used to depict the relative spatial structure (spatial dependence) of spatial data (Ettema and Wardle 2002); spatial structure of 0 indicates that, at a certain sampling scale, the variance of the data is equal to the nugget variance and data points of any separation distance are independent of each other.

We modeled $\gamma(h)$ for each parameter by selecting the best visual fit of all theoretical semivariogram models mentioned above to the semivariance of the data; the gaussian semivariogram model had the best fit for all parameters (Figure 1-4-2). The spatial structure at median plot separation distance, and the sill and range parameters of the respective gaussian models for all data are shown (Table 1-4-2).

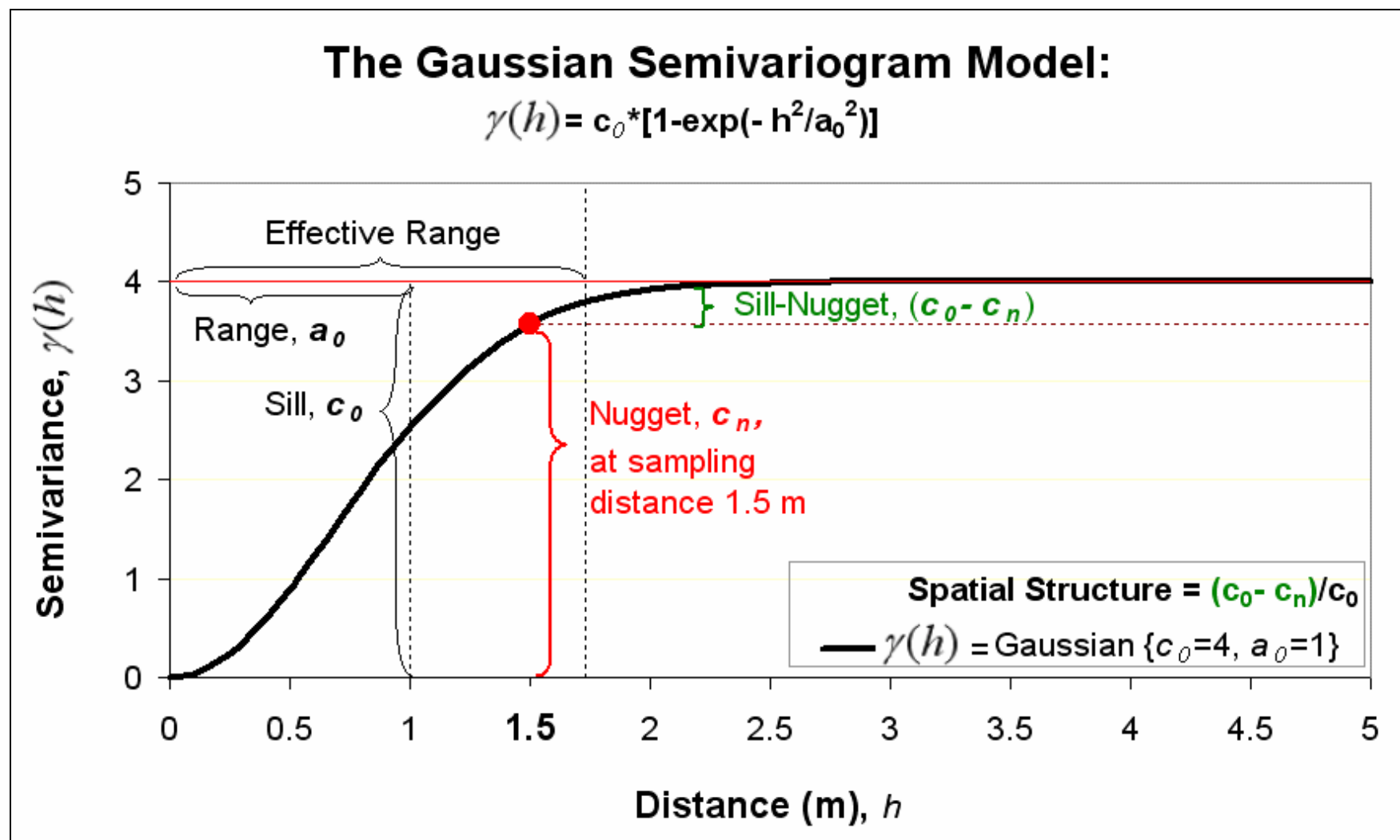


Figure 1-4-2. Components of the gaussian theoretical semivariogram model used to describe the presence or absence of auto-correlation of certain spatial data. The theoretical model with best visual fit to the semivariance statistic of the data was used for spatial data analyses and interpolation, such as kriging (SAS 2007).

Table 1-4-2. Auto-correlation and spatial structure estimates for ecosystem parameters by forest type on mined and adjacent, non-mined land in the Midwestern and Appalachian coalfields of the U.S.

Forest Stand Type	Ecosystem property; C pool	Gaussian model		Spatial structure at median distance	Distance sampled (m)
		c ₀	a ₀		
MINED LAND					
Pine	SI	--	--	--	Min. = 47
	Age	--	--	--	
	Ln(Ecosystem C)	--	--	--	Median = 96
	Ln(Total Tree C)	--	--	--	
	Litter C	--	--	--	Max. = 152
	Sqrt(Soil C)	--	--	--	
Hardwood	SI	2.51	80	0.122	Min. = 33
	Age	--	--	--	
	Ecosystem C	--	--	--	Median = 116
	Total Tree C	--	--	--	
	Litter C	0.85	50	0.005	Max. = 849
	Soil C	--	--	--	
Mixed	SI	--	--	--	Min. = 45
	Age	--	--	--	
	Ln(Ecosystem C)	0.026	60	0.227	Median = 73
	Ln(Total Tree C)	--	--	--	
	Litter C	6.46	60	0.227	Max. = 145
	Sqrt(Soil C)	--	--	--	
NON-MINED LAND					
Hardwood	SI	--	--	--	Min. = 46
	Age	94.00	50	0.048	
	Ln(Ecosystem C)	0.036	60	0.122	Median = 87
	Ln(Total Tree C)	0.024	60	0.122	
	Litter C	--	--	--	Max. = 365
	Ln(Soil C)	--	--	--	

The approximate locations of the sampling plots, in nearly all forest stands (except West Virginia – south), were determined using detailed stand location maps, reported in Rodrigue (2001), and the general location of each stand. Due to the lack of exact latitude and longitude data for each plot, we digitized on-screen the approximate location of each plot using the procedures available in ArcGIS™ software. During the digitizing process, we represented the true distance between closely sampled locations within a certain stand, as opposed to providing the exact latitude and longitude coordinates of a certain sampling point. The separation distance between closely sampled locations is used to estimate the semivariance and the auto-correlation of the data.

The median separation distance between all sampling plots on mined land was 96 m in pine stands, 116 m in hardwood stands, and 73 m in mixed stands; the median distance on non-mined stands was 87 m. We chose analyses at median separation distances, as opposed to the mean distance, because averaging sampling distances would result in a larger overall distance that may not be representative of the entire data set, due to the higher influence of samples separated by large distances to those of shorter distances.

At median sampling distances, the data points in the pine stands were independent; that is, the results from the geostatistical analysis indicated no spatial structure. Similarly for hardwood and mixed mined stands, the results showed no spatial structure for all but two variables (SI and Litter C) in hardwood stands, and all but two variables in mixed stands (Eco_C and Litter C). Site index had a negligible spatial structure indicating that approximately 12% of the variance of the data may be due to auto-correlation of the SI data in the mined hardwood stands (Table 1-4-2). In these stands, less than 1% of the variance of the Litter C data may be due to auto-correlation of the data. In mined mixed stands, approximately 23% of the variance in Eco_C and Litter C may be due to auto-correlation of the respective data from these stands (Table 1-4-2).

Overall, due to the lack of and negligible spatial structure (<23%) for all mined data used in this study, we determined that the mined data by forest type in each plot are statistically independent, and are suitable for the regression analysis used in this study. Similarly, due to the lack of (SI, litter C, and Soil C) and negligible spatial structure (less than 13% for Age, Eco_C, and total tree C) in non-mined stands, we treated these data as independent and suitable for standard statistical analysis.

Identification of Data Outliers

Stand basal area (BA) varied from 95 to 642 thousand-cm² ha⁻¹ along a site index gradient that ranged from 15 to 34 m at stand age 50 yr (Figure 1-4-3). Basal area generally increased with increasing site index, which is an expected response; however, the wide range shows the potential for mined land to be manipulated by the reclamation process. High-quality sites have the capacity to carry higher stocking rates (BA ha⁻¹) and are more productive and sequester more carbon as a result. The variation in the data along a site quality gradient for a given forest type (Figure 1-4-3) is due primarily to factors other than site quality that affect stocking. These factors could include poor early survival, competition from herbaceous vegetation at an early stand age, and mortality caused by disease, herbivory, or physical damage. The effects of these factors are minimized in managed stands.

In order to develop a first approximation model of carbon sequestration as a function of site quality, we wanted to assume that stands represented in our data set were adequately stocked and that differences in basal area within a forest type (pine, mixed, hardwood) would be largely a function of soil and site quality rather than poor survival, unusual mortality, or some other non-site influence on stocking. A check for data outliers was done by regressing stand basal area with site index. Plots with lower-than-expected basal area for a given site index were examined for other potential influences on stocking. Based on observed differences in tree stocking data, we identified several measurement plots as potential data outliers in the data sets for the mined mixed, mined pine, and non-mined hardwood stands.

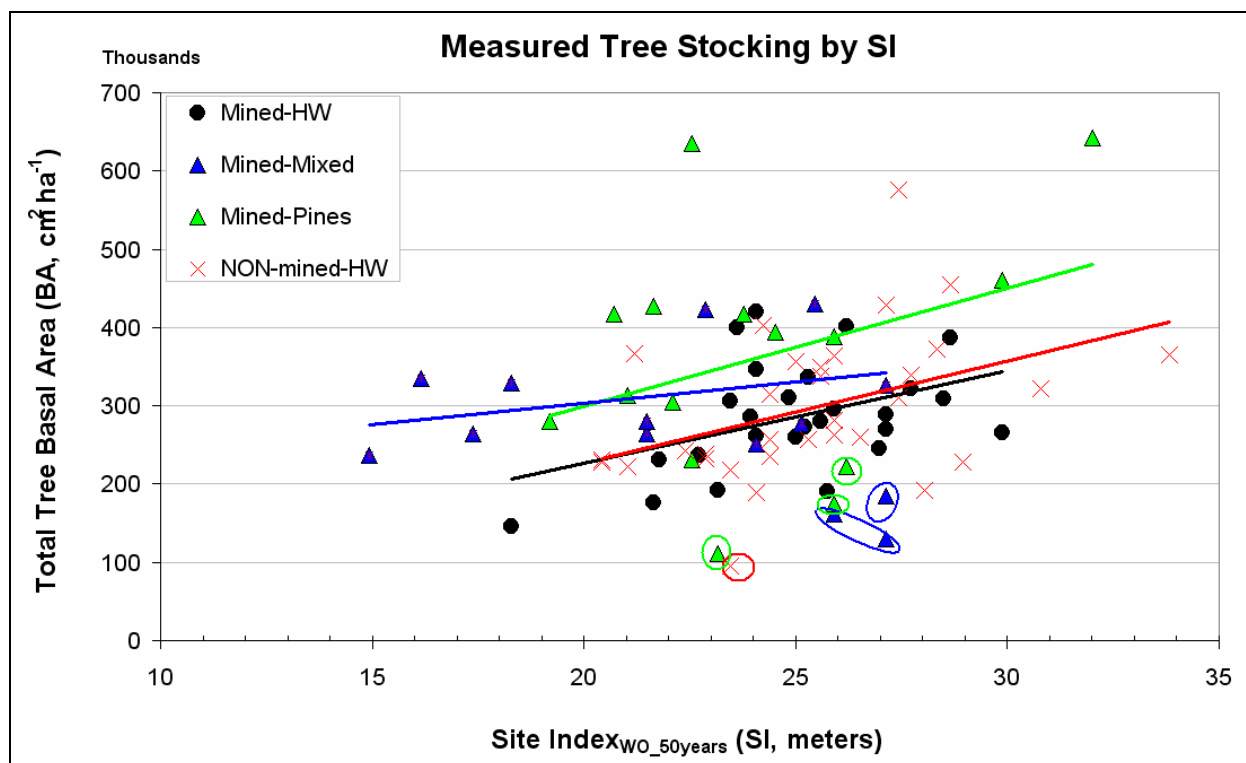


Figure 1-4-3. Basal area (tree stocking) of all measured plots on mined hardwood (Mined-HW), mined mixed, mined pine, and non-mined hardwood (NON-mined-HW) forest stands in the Midwestern and Appalachian coalfields. Regression lines indicate expected trends of stocking with increasing SI. Site index is the height, in meters, for white oak at base age 50 years. Circles indicate the plots identified as data outliers.

The data outliers were IN 1-D1, IN 2-D3, and -F10 plots in Indiana from the mined mixed stand data set; VA-P1, -P2, and -P3 plots in Virginia from the mined pine stand data set; and the PA-C2 plot in Pennsylvania from the non-mined hardwood stand data set (Table 1-4-1, Figure 1-4-3). The BA estimates of these plots were lower than expected and had the greatest deviations from the tree stocking trends developed by SI in their respective data sets (Figure 1-4-3). All these plots were regarded as data outliers and were excluded from all data analyses.

Pitch pine was planted on IN 1-D1, IN 2-D3 and IN 2-F10 plots after mining. Pitch pine was off-site and north of its natural range (Rodrigue 2001). They started collapsing at an early age and yielded to volunteer hardwoods, based on historical records for this site. Rodrigue (2001) indicated that only 61% of the overstory canopy was comprised of pitch pine for the four plots in the IN-1 forest site, and only 46% for the IN-2 site. The rest of the canopy layer was comprised of a large number of small-diameter (DBH<13 cm) hardwood species, which were not included in our analyses. Therefore, we determined that a few large pitch pine trees represented the pitch pine component on IN 1-D1, IN 2-D3, and IN 2-F10 plots, and many small hardwood trees represented the hardwood component.

Our records indicated that the VA-P1, -P2, and -P3 mined plots were located in a white pine stand that had been thinned in 1996, at stand age 17 yr. Although tree growth on these plots was never inhibited or disrupted during the history of the stand, the thinning event compromised the use of these plots. The PA-C2 non-mined plot (Table 1-4-1, Figure 1-4-3) was poorly

drained, which may have caused poor initial stocking of these upland species. The soils on this plot were wet and with a fragipan soil layer that caused a perched water table. There was evidence of impeded drainage between 25 and 44 cm soil depth given the presence of a Btg soil layer.

Two data points in the mined pine data set (plots KY 4-M3 and WV 2-C2) and one from the non-mined hardwood data set (plot PA-A2) appeared to have BA estimates above the respective BA trend lines for the respective stands depicted in Figure 1-4-3. However, our records indicated normal soil and site properties, similar to the rest of the plots within the stands. It is possible that due to the combined effect of factors not accounted for in this analysis, such as superior genotype of the planting stock used on these mined lands (loblolly pine on KY 4-M3, and white pine on WV 2-C2), and a unique microtopography and microclimate could be the reason for higher-than-expected BA levels. Based on the available evidence and despite their elevated BA estimates, these three plots were not treated as data outliers and were used in all analyses.

Ecosystem Carbon Sequestration on Mined and Non-Mined Forest Land

Tree carbon. Little has been done to quantify the carbon pools associated with tree biomass of mature, planted forests on mined sites (Ashby et al. 1980; Burger et al. 2003). Carbon present within the developing tree biomass (stem wood, stem bark, foliage, treetops, branches, stumps, and coarse roots) on our measured non-mined sites ranged from 62 to 233 Mg ha⁻¹ (Table 1-4-3). On mined land, the tree carbon ranged from 62 to 186 Mg ha⁻¹ on pine stands, from 49 to 148 Mg ha⁻¹ on hardwood stands, and 67 to 135 Mg ha⁻¹ on mixed stands. The amount of carbon found in tree biomass was comparable to other forests in the East, considering that tree carbon pools vary from study to study depending on site productivity, site age, tree species, management impacts, and local topography and climate.

Tree carbon averaged 129 Mg ha⁻¹ for the non-mined sites, 116 Mg ha⁻¹ for pine stands, 106 Mg ha⁻¹ for hardwood stands, and 91 Mg ha⁻¹ for mixed stands on the mined sites (Table 1-4-3). Carbon associated with tree biomass of hardwood forests in Indiana were measured by Kaczmarek et al. (1995). For all but four non-mined and mined hardwood plots in Indiana and western Kentucky (Table 1-4-3), our estimates were well within the range of their carbon estimates, which were between 61 and 117 Mg ha⁻¹.

Richter et al. (1995) found 140.6 Mg C ha⁻¹ in the tree biomass of a 35-year-old loblolly pine site in South Carolina. Within our study, sites KY-3 and KY-4 both contained loblolly pine. The tree carbon on these plots ranged from 99 to 170 Mg ha⁻¹ of carbon in total woody plant biomass at stand ages ranging from 30 to 38 years. Mixed stands in the Appalachian region, in Indiana, West Virginia (north), and West Virginia (south) had tree carbon content averaging 77, 81, and 118 Mg ha⁻¹, respectively (Table 1-4-3). Carbon stocks in these mixed stands compared favorably with carbon estimates for mixed pine and hardwood canopies on southern and western aspects in North Carolina (Vose and Swank 1993).

Table 1-4-3. Carbon sequestered by ecosystem component at present age and projected to age 60 years in even-aged stands on 55 mined and 32 non-mined (adjacent) forested plots. Shaded rows indicate measurement plots identified as outliers. Average values were estimated without outliers. The Walkley-Black (WB) method was used to measure the soil carbon on non-mined land. Coal-corrected SOC and the original WB soil carbon values are presented for mined land.

Forest Type	State / Site	Total Tree Carbon	Soil Carbon		Litter Layer Carbon	Ecosystem Carbon	Projected Ecosystem C at 60 years
			SOC	WB-C*			
----- Mg ha ⁻¹ -----							
<u>MINED LAND</u>							
Pine stands	IN 1 - B7	118.2 (71) ‡	13.5 (8) ‡	35.8	35.1 (21) ‡	166.8	160.6
	KY 3 - B2	99.4 (80)	15.9 (13)	49.3	8.6 (7)	123.8	133.3
	KY 4 - L1	113.1 (74)	12.6 (8)	40.0	27.9 (18)	153.6	189.3
	KY 4 - M3	170.3 (82)	19.3 (9)	57.7	18.8 (9)	208.4	249.0
	KY 4 - X2	108.1 (77)	8.5 (6)	24.2	24.6 (17)	141.3	161.0
	Average	122.7 (78)	14.1 (9)	42.8	20 (13)	156.8	183.2
	PA 1 - A4	116.2 (81)	4.5 (3)	12.1	22.8 (16)	143.5	110.2
	PA 1 - C5	77 (70)	8.6 (8)	24.8	24.8 (22)	110.5	113.4
	PA 1 - B6	62 (78)	2.5 (3)	7.6	15 (19)	79.6	79.8
	PA 1 - F2	92.5 (81)	7.2 (6)	21.5	14.5 (13)	114.2	105.0
	Average	86.9 (78)	5.7 (5)	16.5	19.3 (17)	111.9	102.1
	VA 1 - P1	27.6	n/a	n/a	14.0	n/a	n/a
	VA 1 - P2	57.2	n/a	n/a	19.1	n/a	n/a
	VA 1 - P3	44.4	n/a	n/a	15.0	n/a	n/a
	Average	43.1	n/a	n/a	16.0	n/a	n/a
	WV 2 - C4	186.2 (84)	21.5 (10)	66.8	15 (7)	222.7	317.2
	WV 2 - D2	115.6 (85)	3.4 (3)	10.5	17.3 (13)	136.3	157.1
	WV 2 - G2	135 (75)	14.8 (8)	43.9	29.8 (17)	179.7	244.7
	Average	145.6 (81)	13.2 (7)	40.4	20.7 (12)	179.6	239.7
	Average		116.1 (78)	11 (7)	32.9	21.2 (14)	148.4
Mixed stands	IN 1 - D1	46.5 (61)	8.4 (11)	21.9	20.8 (27)	75.7	65.2
	IN 1 - E3	72.9 (74)	8.9 (9)	26.0	17.2 (17)	98.9	93.8
	IN 1 - G2	84.2 (77)	14.5 (13)	42.8	10 (9)	108.7	111.2
	IN 2 - A9	74.9 (74)	14.4 (14)	41.9	12.1 (12)	101.4	104.2
	IN 2 - D3	37.2 (60)	13.9 (22)	36.3	10.7 (17)	61.8	75.0
	IN 2 - F10	61.6 (55)	30.9 (27)	75.2	20 (18)	112.6	127.7
	Average	77.3 (75)	12.6 (12)	36.9	13.1 (13)	103.0	103.1
	KY 3 - A4	99.3 (73)	31.7 (23)	80.8	5.5 (4)	136.5	175.8
	KY 3 - C3	134.8 (85)	15.2 (10)	48.2	8.7 (5)	158.6	173.4
	KY 3 - D1	90 (79)	17 (15)	43.6	7.2 (6)	114.2	121.4
	Average	108 (79)	21.3 (16)	57.6	7.1 (5)	136.4	156.9
	WV 1 - B3	93.4 (79)	4.3 (4)	13.8	21 (18)	118.7	109.0
WV 1 - D1	91.1 (81)	1.4 (1)	4.4	19.8 (18)	112.4	123.8	
WV 1 - E3	67.4 (79)	5.1 (6)	15.6	13 (15)	85.5	83.1	
WV 1 - F3	70.4 (71)	9.2 (9)	28.4	19.5 (20)	99.1	118.7	
Average	80.6 (78)	5 (5)	15.6	18.3 (18)	103.9	108.7	
WV 2 - A2	118.1 (74)	21.6 (14)	70.7	19.1 (12)	158.8	227.6	
Average		90.6 (77)	13 (11)	37.9	13.9 (12)	117.5	131.1

Table 1-4-3 (continued).

Table 1-15 (Continued).							
Forest Type	State / Site	Total Tree Carbon	Soil Carbon		Litter Layer Carbon	Ecosystem Carbon	Projected Ecosystem C at 60 years
			SOC	WB-C*			
----- Mg ha ⁻¹ -----							
Hardwood stands	IL 1 - B1	128.3 (81) ‡	21.9 (14) ‡	55.9	7.9 (5) ‡	158.1	170.8
	IL 1 - E2	131.3 (83)	20.2 (13)	54.2	6.2 (4)	157.7	173.3
	IL 1 - G2	144.7 (86)	16.8 (10)	45.1	6.6 (4)	168.1	178.2
	IL 1 - I1	108 (86)	12.6 (10)	37.5	5.6 (4)	126.2	141.4
	IL 2 - A2	112.9 (82)	18.7 (14)	57.0	6.8 (5)	138.4	151.8
	IL 2 - C2	85.1 (80)	12.9 (12)	39.3	8.3 (8)	106.4	120.8
	IL 2 - E1	147.7 (89)	9.6 (6)	32.1	8.2 (5)	165.6	181.4
	IL 2 - G4	140.6 (88)	12.3 (8)	40.4	7.2 (4)	160.1	180.5
	Average	124.8 (85)	15.6 (11)	45.2	7.1 (5)	147.6	162.3
	IN 2 - E6	48.8 (55)	31.1 (35)	79.3	8.6 (10)	88.5	158.5
	KY 1 - B4	74.7 (78)	12.3 (13)	40.5	8.2 (9)	95.1	132.6
	KY 1 - F2	56.8 (76)	10.9 (15)	33.4	6.6 (9)	74.2	104.6
	KY 1 - G1	114.4 (75)	31.1 (20)	91.7	7.6 (5)	153.0	215.5
	KY 1 - H2	70.5 (66)	28.5 (27)	72.3	7.6 (7)	106.5	154.5
	KY 1 - J2	64 (73)	14.2 (16)	45.9	8.9 (10)	87.1	121.7
	KY 2 - A5	72.3 (83)	10.6 (12)	34.4	4 (5)	86.9	108.4
	KY 2 - G5	96.2 (78)	24.7 (20)	80.5	2.8 (2)	123.6	159.2
	KY 2 - E4	110.5 (88)	10.7 (8)	34.1	4.6 (4)	125.9	147.1
	KY 2 - K1	93 (88)	8.6 (8)	26.8	3.7 (4)	105.3	124.0
	Average	83.6 (79)	16.8 (16)	51.1	6 (6)	106.4	140.8
	OH 1 - A1	135.2 (77)	36.9 (21)	109.3	3.6 (2)	175.7	204.3
	OH 1 - B3	121.6 (78)	28.9 (19)	76.3	4.7 (3)	155.2	170.7
	OH 1 - D4	100.6 (82)	17.8 (15)	43.8	3.7 (3)	122.2	136.3
	OH 1 - F5	143.5 (87)	15.9 (10)	37.8	5.8 (4)	165.2	170.8
	OH 2 - B7	137.5 (89)	10.7 (7)	34.0	5.6 (4)	153.8	142.3
	OH 2 - B2	98.5 (87)	10.1 (9)	31.7	4.2 (4)	112.8	144.9
	OH 2 - C3	117.7 (92)	4.4 (3)	13.1	5.3 (4)	127.4	134.9
	OH 2 - D4	103.9 (75)	28.7 (21)	77.9	5.9 (4)	138.5	156.3
	Average	119.8 (83)	19.2 (13)	53.0	4.9 (3)	143.8	157.6
Average	106.1 (82)	17.7 (14)	50.9	6.1 (5)	129.9	153.3	
<u>NON-MINED LAND (adjacent to mined sites)</u>							
Hardwood stands	IL - A2	176.9 (56) ‡	131.5 (42) ‡		6.5 (2) ‡	314.9	327.8
	IL - B9	134.1 (46)	148 (51)		6.4 (2)	288.5	294.2
	IL - G4	176.2 (74)	55.8 (23)		6.2 (3)	238.2	254.1
	IL - F8	137.9 (57)	97.3 (40)		8.6 (4)	243.8	254.8
	Average	156.3 (58)	108.1 (40)		6.9 (3)	271.3	282.7
	IN - B1	80.9 (47)	87.8 (51)		3.7 (2)	172.4	190.2
	IN - C4	88.5 (55)	66.8 (42)		5.1 (3)	160.4	166.8
	IN - D1	198.9 (70)	79.2 (28)		7.7 (3)	285.7	284.6
	IN - D6	84.2 (56)	60.4 (40)		5.2 (3)	149.8	153.7
	Average	113.1 (59)	73.5 (38)		5.4 (3)	192.1	198.8

Table 1-4-3 (continued).

Forest Type	State / Site	Total Tree Carbon	Soil Carbon		Litter Layer Carbon	Ecosystem Carbon	Projected Ecosystem C at 60 years
			SOC	WB-C*			
----- Mg ha ⁻¹ -----							
Hardwood stands	KY - C	145.3 (63) ‡	77.9 (34) ‡		9.2 (4) ‡	232.3	241.9
	KY - E	91.2 (59)	52.9 (34)		11.4 (7)	155.6	193.4
	KY - J	72.9 (44)	79.4 (48)		14.5 (9)	166.8	177.6
	KY - O	97.3 (60)	55.1 (34)		10.8 (7)	163.2	172.0
	Average	101.7 (57)	66.3 (37)		11.5 (6)	179.5	196.2
	OH - A3	94 (57)	61.8 (37)		9.7 (6)	165.5	167.3
	OH - C10	95.4 (60)	53.9 (34)		10.3 (6)	159.7	166.1
	OH - C4	94.7 (66)	42.3 (30)		6.3 (4)	143.3	145.8
	OH - E3	105.3 (62)	55.1 (33)		9.1 (5)	169.5	164.4
	Average	97.3 (61)	53.3 (33)		8.9 (6)	159.5	160.9
	PA - A2	233.3 (81)	43.3 (15)		12.9 (4)	289.5	284.3
	PA - C2	34.7 (35)	52.3 (53)		11.7 (12)	98.7	99.7
	PA - G2	114.2 (62)	52.5 (29)		17.1 (9)	183.7	187.4
	PA - J1	96.9 (61)	49.1 (31)		11.9 (8)	157.9	154.8
	Average	148.1 (70)	48.3 (23)		14 (7)	210.4	208.8
	VA - A2	171.4 (68)	62 (25)		17 (7)	250.3	233.6
	VA - B6	165.5 (68)	64.5 (27)		12.1 (5)	242.1	239.5
	VA - C7	116.1 (60)	64.4 (33)		13.4 (7)	193.9	185.7
	VA - E1	140.2 (60)	75.1 (32)		17.6 (8)	232.9	230.7
	Average	148.3 (65)	66.5 (29)		15 (7)	229.8	222.4
	WV 1 - A2	134.5 (66)	60.3 (30)		9 (4)	203.8	209.0
	WV 1 - B4	118.8 (56)	80.1 (38)		11.6 (6)	210.5	205.9
	WV 1 - D4	137.1 (69)	54 (27)		7.3 (4)	198.4	192.7
	WV 1 - E4	137.4 (55)	96.8 (39)		13.5 (5)	247.7	245.6
	Average	131.9 (61)	72.8 (34)		10.3 (5)	215.1	213.3
	WV 2 - A6	130.6 (61)	74.9 (35)		7 (3)	212.6	216.6
	WV 2 - B1	62 (30)	137.3 (67)		6.1 (3)	205.4	217.0
	WV 2 - C2	150.9 (66)	68.7 (30)		10.5 (5)	230.1	228.1
	WV 2 - D5	203.6 (80)	44.3 (17)		8.1 (3)	256.1	265.9
	Average	136.8 (61)	81.3 (36)		8 (4)	226.0	231.9
Average		128.6 (61)	72 (34)		9.9 (5)	210.5	205.4

* WB-C, Walkley-Black estimate of soil carbon

‡ Numbers in parenthesis (carbon values) represent carbon distribution as percent of total ecosystem carbon. Sum of percent values may be greater or smaller than 100 due to rounding.

** Projected ecosystem carbon estimates at age 60 years are the sum of the projected tree, soil, and litter layer C pools as described in the text.

Overall, our mined study sites were new forest communities that, in some cases, accumulated nearly as much carbon in the tree biomass as that sequestered in trees on the non-mined sites (Table 1-4-3). Most of these new forests contained valuable species and trees were well spaced (Rodrigue 2001; Rodrigue et al. 2002). Our data showed that low-quality sites can be more productive after mining with the potential to develop greater amounts of tree carbon per unit area, especially for stands that are understocked, high-graded forests, commonly found in the eastern United States.

Litter layer carbon. Non-mined-stand litter layer carbon pools ranged from 4 to 18 Mg C ha⁻¹ (Table 1-4-3). Litter layer carbon pools on mined land ranged from 9 to 35 Mg C ha⁻¹ in pine stands, from 3 to 9 Mg C ha⁻¹ in hardwood stands, and from 6 to 21 Mg C ha⁻¹ in mixed stands (Table 1-4-3).

Litter layer carbon estimates compared favorably with other investigations. Overall, the litter layer carbon estimates from non-mined sites averaged 10 Mg C ha⁻¹ and those from mined sites averaged 21 Mg C ha⁻¹ in pine stands, 6 Mg C ha⁻¹ in hardwood stands, and 14 Mg C ha⁻¹ in mixed stands. Litter layer carbon pools from other studies in the eastern U.S. ranged from 4 to 14.4 Mg C ha⁻¹ depending on age and forest species (Hoover et al. 2000; Kaczmarek et al. 1995; Van Lear et al. 1995; Vose and Swank 1993). However, sites with a higher conifer component tended to develop greater carbon pools within the litter layer. In Ohio, Vimmerstedt et al. (1989) found that hardwood litter layer carbon pools (3 Mg ha⁻¹) were significantly lower than litter layer carbon pools on pine sites (8 Mg ha⁻¹). On a 35-year-old loblolly pine site in the Piedmont of South Carolina, litter layer carbon averaged 32.8 Mg C ha⁻¹ (Richter et al., 1995). The litter layer carbon on our loblolly pine plots in western Kentucky (KY-4), with stand age ranging from 30 to 35, averaged 24 Mg C ha⁻¹ (Table 1-4-3). The litter layer input/output dynamic may be at or near equilibrium in most of our measured stands.

Soil carbon. Unlike tree and litter layer carbon, soil carbon content on mined sites was consistently lower than that of the non-mined soil carbon pool. Non-mined-site soil carbon ranged from 42 to 137 Mg ha⁻¹ (Table 1-4-3). The amount of carbon in the mine soils ranged from 2 to 22 Mg ha⁻¹ under pines, from 4 to 37 Mg ha⁻¹ under hardwoods, and from 1 to 32 Mg ha⁻¹ under mixed forest vegetation.

Average non-mined-site soil carbon levels were similar to estimates for temperate forest soil carbon levels reported in the literature. Post et al. (1982) reported average soil carbon levels of 79 and 60 Mg ha⁻¹ within 1 m for dry and moist warm temperate forests, respectively. Researchers in the eastern U.S. reported carbon levels for depths from 0.5 m to bedrock ranging between 36 and 130 Mg ha⁻¹ (Hoover et al. 2000; Johnson et al. 1995; Kaczmarek et al. 1995). The non-mined sites in our study averaged 72 Mg C ha⁻¹, and the mined sites averaged 11 Mg C ha⁻¹ in pine stands, 18 Mg C ha⁻¹ in hardwood stands, and 13 Mg C ha⁻¹ in mixed stands (Table 1-4-3).

Our mined site average soil carbon estimates were lower than carbon pools reported in other investigations. Akala and Lal (1999) reported that 30-year-old reforested mined sites (to 0.5 m depth) contained 51.5 Mg ha⁻¹ of soil carbon, and 50-year-old sites contained 54.9 Mg C ha⁻¹. Mined sites approximately 30 years old in our study contained 16 Mg C ha⁻¹, while approximately 50-year-old mined sites contained an average of 17 Mg C ha⁻¹ (Table 1-4-3). However, in a later study, Akala and Lal (2001b) mentioned the possibility of reporting incorrectly measured soil carbon estimates, due to the measurement technique used, noting that

“contamination due to coal and shale can lead to erroneous measurements in SOC content.”

Akala and Lal (2001b) used the Walkley-Black procedure to measure SOC on their mined study sites. The average Walkley-Black soil carbon estimate for mined sites in the 25- to 35-year and 45- to 55-year age categories in our study was approximately 46 Mg C ha⁻¹ (Table 1-4-3).

Results from our previous experiments to evaluate the performance of the Walkley-Black procedure indicated that the Walkley-Black method could produce erroneous SOC results when used for mine soil analysis due to coal particles that are present in most mined sites (Substudy 1). The original, uncorrected Walkley-Black carbon values, and the coal-corrected SOC estimates for the mined sites in this study are reported in Table 1-4-3. On average, the SOC over-estimation of Walkley-Black carbon values ranged from 138% (in Ohio) to 236% (in Illinois) (Table 1-4-3). The correction equation, described in the Methods section of this substudy, was based on data from carefully constructed and homogeneously mixed mine soil samples of a known amount of coal material. The consistency among our corrected SOC results supported the argument that Walkley-Black carbon data sets for mined land compiled in the past, similar to those reported by Rodrigue (2001) and Akala and Lal (2001b), can be successfully corrected for coal content.

Of the three ecosystem components, the soil had the greatest potential for sequestering and storing additional carbon. Mine soils could be regarded as “empty cups” that could accumulate and store large amounts of sequestered carbon. Assuming that hardwood forest would be the long-term (decades) land use practice of a certain mined site, the current mine soil organic carbon levels would increase considerably until they reach the equilibrium SOC levels approximating natural hardwood forests. The results showed that mine soils under hardwood vegetation have the potential to store 307% more carbon, resulting in a more than three-fold increase from 18 Mg C ha⁻¹ (mined hardwood stands) to 72 Mg C ha⁻¹ (non-mined hardwood stands), provided the mined land is managed as forest (Table 1-4-3).

The soil carbon pools on the mined lands had not reached equilibrium state (Figure 1-4-4). Assuming that soil carbon would accumulate at a constant rate, it would take about 85 years (ranging between 64 years in the stands in Virginia and 116 years for mined sites in Illinois) for these mined sites to reach the soil carbon equilibrium levels of the adjacent, non-mined forests. These estimates supported the findings of Akala and Lal (2001b), who reported that reclaimed forest land in Illinois could attain soil carbon equilibrium between 110 and 150 years following forest establishment.

Using the same land-use assumption as above, the data indicated that the increase in the amounts of sequestered carbon in the litter layer and tree biomass of a mined hardwood forest would be approximately one order of magnitude lower than that of the soil component, provided mined hardwood stands remain as forest for a few decades into the future. Twenty-two-percent (from 106 to 129 Mg C ha⁻¹) and 62% (from 6.1 to 9.9 Mg C ha⁻¹), respectively, would be the expected long-term increases in the amount of sequestered carbon in tree biomass and litter layer on mined land under hardwood forest vegetation (Table 1-4-3).

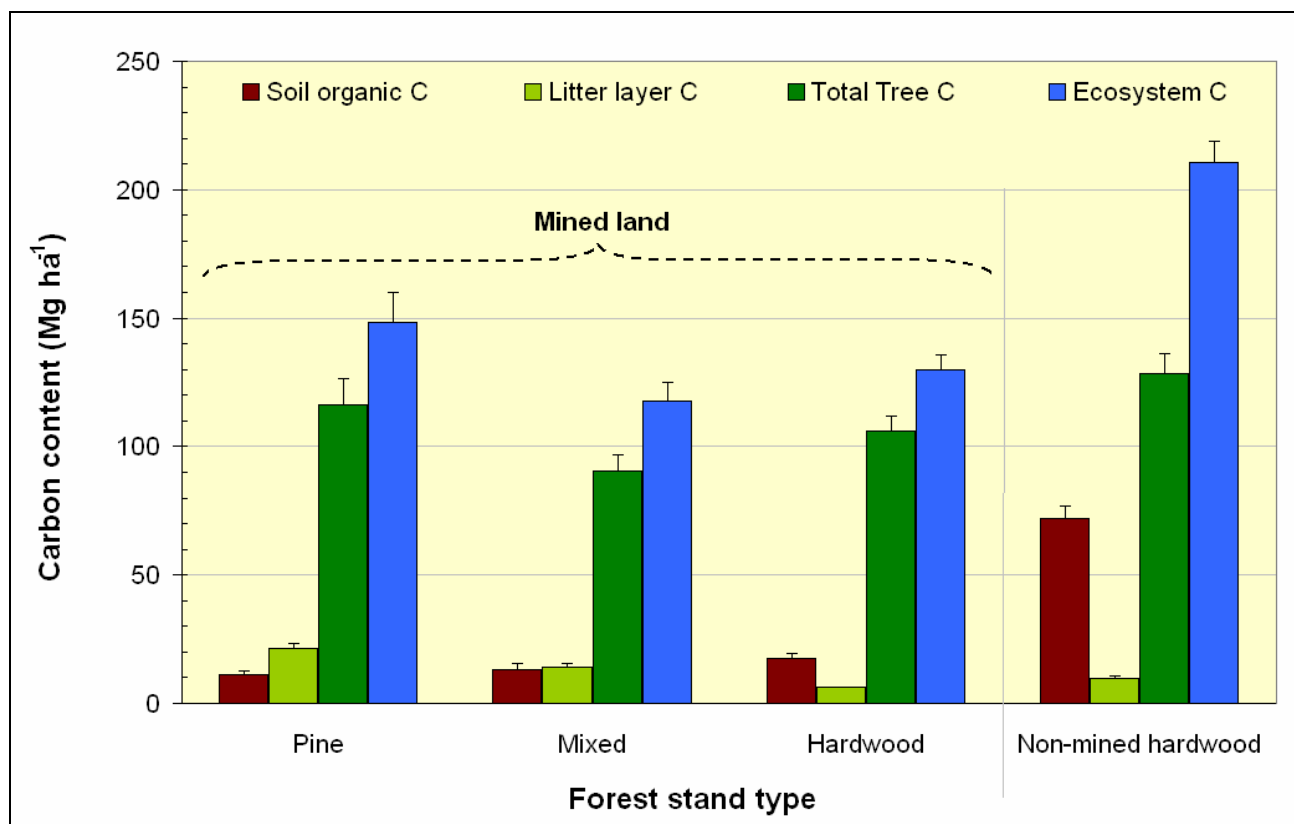


Figure 1-4-4. Carbon pools by ecosystem component, soil, litter layer, tree, and total ecosystem carbon content, for three forest stand types on mined land, pine, mixed, hardwood, and for non-mined hardwood stands, adjacent to the mined stands. Vertical lines represent one standard error of the mean estimate.

Total ecosystem carbon. On average, the highest amount of ecosystem carbon on these forested mined sites was sequestered in pine stands (148 Mg ha^{-1}), followed by hardwood stands (130 Mg ha^{-1}) and mixed stands (118 Mg ha^{-1}) (Fig. 1-4-4, Table 1-4-3). As expected, more than three-quarters of the total terrestrial carbon on mined land was sequestered in the form of tree biomass (including above-ground biomass and coarse roots): 78% in pine stands, 82% in hardwood stands, and 77% in mixed stands (Table 1-4-3).

The distribution of the remaining one-quarter of the total ecosystem carbon among the soil and litter layer components differed by forest type on the mined study sites. Fourteen percent of the ecosystem carbon in pine stands was located in the litter layer, and half that amount, 7%, was in the soil. In hardwood stands, the distribution of carbon between litter layer and soil was reversed: 14% was measured in the soil and 5% was in the litter layer. In the mixed stands, the distribution was about equal (Fig. 1-4-4, Table 1-4-3). Litter decomposition rates differ with litter type, a result of variation of litter quality, litter chemical composition, availability of nutrients, particularly nitrogen, from other site resources, and climatic factors (Smith and Heath 2002; Vogt et al. 1986). Because of the different chemical composition of the foliage in pine stands, particularly the higher lignin and wax content, different leaf structure, and more acidic conditions, compared to the foliage under hardwood tree canopy, litter layers in pine stands tend to accumulate due to slower rates of decomposition. Therefore, less decomposed organic matter is released from the litter layer to move into the soil.

In the long term, as soil carbon continues to accumulate in forested mined land, the carbon distribution between litter and soil will change, but changes will probably occur at different rates under pine and hardwood vegetation as suggested by the different amounts of carbon in the various pools in these stands. For the non-mined study sites, 34% of the total ecosystem carbon content (210 Mg C ha^{-1}) was found in the soil, 61% in tree biomass, and 5% in litter layer (Table 1-4-3).

Non-mined hardwood stands contained 210 Mg C ha^{-1} , which was about 62% higher than the average of all mined stands (130 Mg C ha^{-1}). On average, the non-mined hardwood stands had approximately 42%, 62%, and 79% more cumulative carbon in tree biomass, litter layer, and soils, compared to mined pine, mined hardwood, and mined mixed forest stands, respectively (Table 1-4-3, Fig. 1-4-4). With time, each ecosystem component will reach its carbon carrying capacity, with the soil component lagging the litter component, and the tree component lagging both the soil and litter components.

Based on the projected ecosystem carbon estimates in Table 1-4-3, the average rotation age ecosystem carbon content in non-mined stands was 205 Mg C ha^{-1} , which was about 34% greater than the average for all mined stands. Among the mined pine stands measured in four states (excluding the data outliers in Virginia), the rotation age ecosystem carbon content was in the following decreasing order, West Virginia (south) > Kentucky > Indiana > Pennsylvania, and ranged from 102 to 240 Mg C ha^{-1} (Table 1-4-3). In mined hardwood stands, the rotation age ecosystem carbon content ranged from 141 to 162 Mg C ha^{-1} in the decreasing order (by state) Illinois > Indiana > Ohio > Kentucky. Similarly, in mined mixed stands, the rotation age ecosystem carbon content ranged from 103 to 228 Mg C ha^{-1} in the decreasing order (by state) West Virginia (south) > Kentucky > West Virginia (north) > Indiana (Table 1-4-3).

Effects of Mine Site Quality and Stand Age on the Carbon Sequestration Potential of Forested Mined Land

Site index is often used as a measure of site quality on land that can support tree growth and is currently a forest (Avery and Burkhart, 1994a). Site index is the height of a dominant or a co-dominant tree within a forest stand at a certain base age; e.g., age 50 for white oak, and age 25 for white pine. The site indices of our study plots were reported for white oak at age 50 (Rodrigue 2001) because white oak is native on all our sites, and it is an important commercial species in the Midwestern and Appalachian regions.

For nearly all mined sites, tree and ecosystem carbon content were correlated with site index and stand age, largely because 75% of the total carbon was in tree biomass (Table 1-4-4). Significant correlation ($P < 0.10$) between the ecosystem carbon pool and the ratio of SI over age, depicting the interaction between SI and age, indicated that a potential response surface from a regression model could be established that would describe ecosystem carbon sequestration using SI and age as independent variables.

Table 1-4-4. Pearson correlation coefficients for tree, litter, soil, and total ecosystem carbon content, as the dependent variables, by SI, Age, SI/Age, as the independent variables, for three forest types on mined land in the Midwestern and Appalachian coalfields. Values in parenthesis indicate the probability (P-value) for the correlation coefficient to be equal to zero. Shaded cells represent significant linear relationship between the dependent (transformed measurements) and independent variables at the 90% probability level. Values in bold type indicate inverse linear relationships. Site index (SI) is the height, in meters, for white oak at base age 50 years.

Forest stand type	C pool, Mg C ha ⁻¹	Variable transformation*	Independent variables			n	
			SI	Age	SI/Age		
----- correlation coefficient & (P-value) -----							
<u>Pine</u>	Tree	<i>Ln</i>	0.6277 (0.0289)	-0.5217 (0.082)	0.6466 (0.0231)	12	
	Litter	n/a	0.0573 (0.8596)	0.0592 (0.8551)	0.0215 (0.9472)	12	
	Soil	<i>SQRT</i>	0.4250 (0.1684)	-0.3847 (0.2169)	0.4627 (0.1299)	12	
	<i>Ecosystem</i>	<i>Ln</i>	0.6014 (0.0386)	-0.4974 (0.0999)	0.6226 (0.0306)	12	
	Hardwood	Tree	n/a	0.6164 (0.0008)	0.6211 (0.0007)	-0.4162 (0.0344)	26
<u>Hardwood</u>	Litter	n/a	-0.3606 (0.0703)	-0.1249 (0.5431)	-0.0435 (0.8329)	26	
	Soil	<i>SQRT</i>	-0.2755 (0.1731)	-0.0935 (0.6497)	-0.0755 (0.714)	26	
	<i>Ecosystem</i>	n/a	0.4944 (0.0102)	0.5675 (0.0025)	-0.4294 (0.0286)	26	
	Mixed	Tree	<i>Ln</i>	0.5264 (0.0962)	-0.3146 (0.346)	0.6487 (0.0308)	11
	Litter	n/a	-0.7124 (0.0139)	-0.3692 (0.2639)	-0.2651 (0.4308)	11	
<u>Mixed</u>	Soil	<i>SQRT</i>	0.8347 (0.0014)	-0.0477 (0.8893)	0.7299 (0.0108)	11	
	<i>Ecosystem</i>	<i>Ln</i>	0.5928 (0.0546)	-0.3920 (0.2331)	0.7705 (0.0055)	11	

* Dependent variable transformation required to achieve normal distribution of the data:
n/a - no transformation required; *Ln* - natural log; *SQRT* - square root;

Site index and age were correlated differently with the tree, litter, soil, and ecosystem carbon pools for the mined sites in this study (Table 1-4-4). We expected the carbon content in all carbon pools to increase with the increase in site quality (i.e., higher SI) because, as trees capture atmospheric CO₂, more carbon is deposited on the soil surface as litter layer biomass (foliage, branches), or in the soil as soil organic matter from decomposing roots and foliage, and root exudates. Site index was positively correlated with the tree carbon pool for all mined sites (Table 1-4-4). In contrast, the litter layer carbon pools for the three forest types were either not correlated (pine stands) or were negatively correlated with SI (hardwood and mixed stands). Neither the litter layer nor soil carbon pools were correlated with stand age. Soil carbon content was correlated with SI only under mixed forest vegetation and there was no significant relationship between SI and soil carbon in pine and hardwood stands (Table 1-4-4).

In pine stands, the tree carbon pool was significantly correlated with SI ($P = 0.0289$), age ($P = 0.0820$), and with the ratio of SI over age ($P = 0.0231$). Changes in litter layer and soil carbon content under pine vegetation appeared to be unrelated to changes in SI ($P > 0.1$), age ($P > 0.1$), or the interaction between SI and age ($P > 0.1$) (Table 1-4-4). As expected, SI was positively correlated with tree carbon content (Pearson correlation coefficient, $r = 0.6277$), indicating that mined land of higher site quality would sequester significantly higher amounts of carbon in tree biomass compared to low quality sites.

Similarly, the results indicated that SI and total ecosystem carbon content under pine vegetation were positively correlated ($P = 0.0386$). The correlation between stand age and the amounts of sequestered tree and ecosystem carbon in pine stands was negative, r equal to -0.5217 and -0.4974, respectively (Table 1-4-4). This finding followed our observation that forest stands with a high, planted loblolly or pitch pine component began to collapse and be replaced with native hardwoods beginning at approximately age 30. The age of the pine stands in this study ranged from 27 to 46 years. The decrease in ecosystem carbon with stand age in pine stands older than 25 years was also depicted by the response surface for ecosystem carbon under pine vegetation for the mined sites (Figure 1-4-5). Ecosystem carbon content for pine stands increased exponentially with the increase of SI.

Ecosystem carbon response surfaces allowed inferences to be made regarding the carbon sequestration potential of mined land supporting different types of forest vegetation (Figs. 1-4-5 and 1-4-6). The response surface with the best-fit to the data was developed for the mined mixed data set, which explained 59% of the variation in ecosystem carbon across the spectrum of SI (ranging from 10 to 35 m) and stand age (ranging from 25 to 60 years) (Figure 1-4-6). The regression models for the mined pine and mined hardwood stands accounted for 39 and 35%, respectively, of the variation in ecosystem carbon content across the spectrum of SI and Age (Figures 1-4-5 and 1-4-6).

The tree ($P = 0.0008$), litter ($P = 0.0703$), and ecosystem ($P = 0.0102$) carbon pools in mined hardwood stands were significantly correlated with changes in site index. However, SI and litter carbon were negatively correlated, in contrast to the common expectation that better sites that could support higher tree biomass should also store larger amounts of sequestered carbon in the litter layer. We believe that due to relatively better soil conditions on high quality sites, decomposition rates of litter biomass are much greater than those on low quality sites under hardwood vegetation. Therefore, at the current young stages of mine soil development and establishment of microbial communities, litter layer biomass is decomposed faster on high quality mined sites, compared to low quality sites.

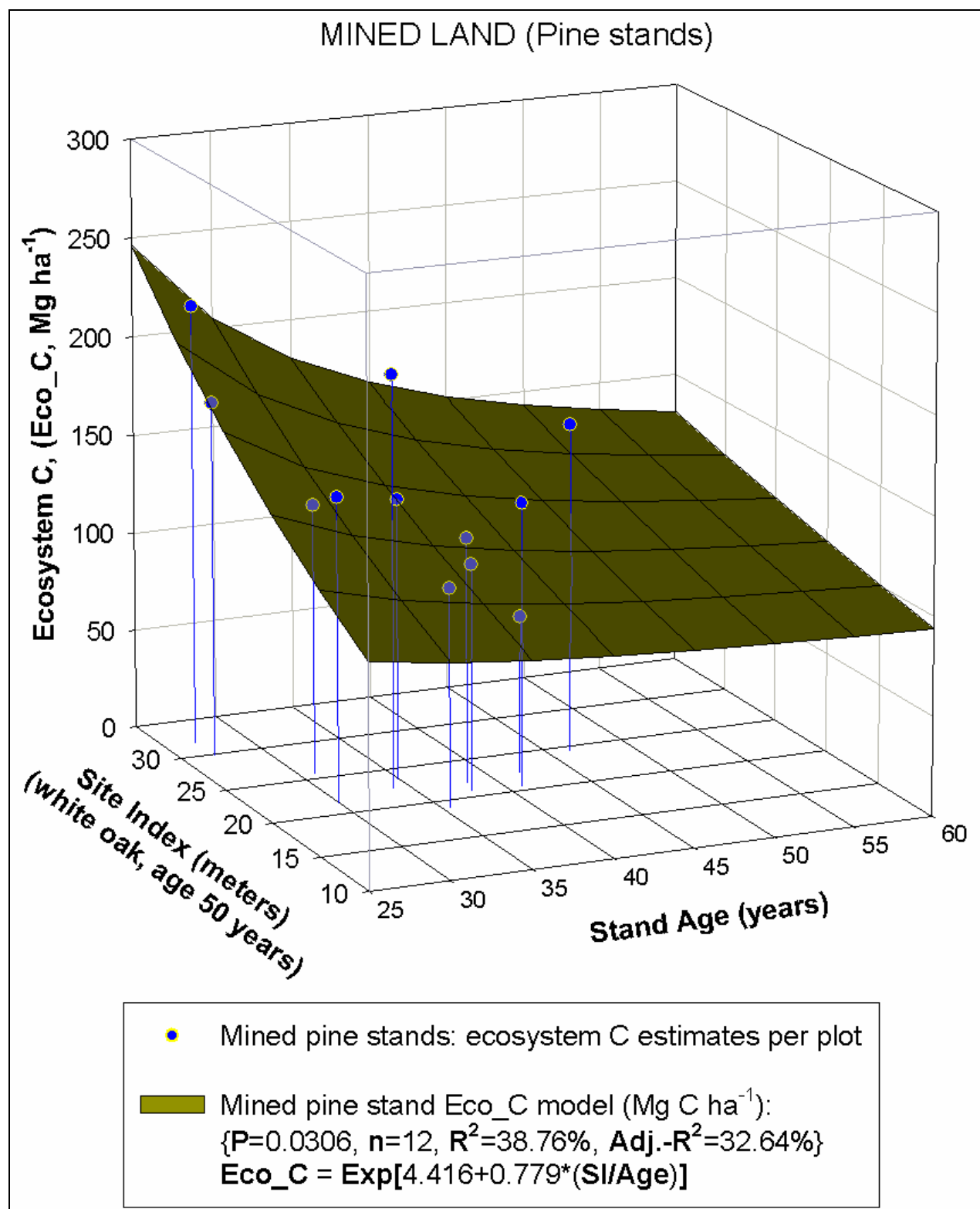


Figure 1-4-5. Ecosystem carbon content response surface by site index and stand age for pine stands on mined land in the Midwestern and Appalachian coal fields. Site index is the height for white oak at base age 50 years.

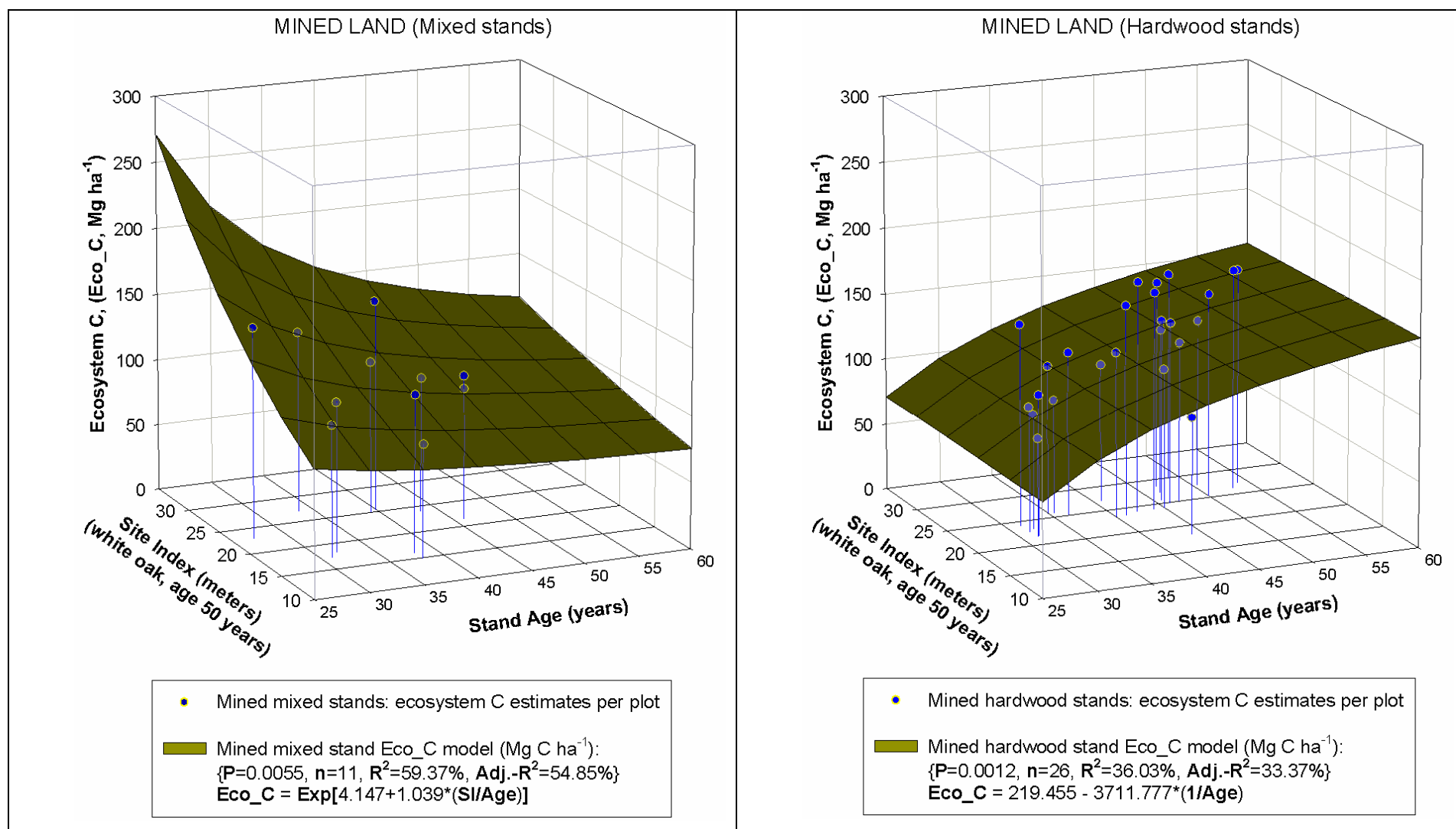


Figure 1-4-6. Ecosystem carbon content response surfaces by site index and stand age for mixed (left) and hardwood (right) forest stands on mined land in the Midwestern and Appalachian coalfields. Site index is the height for white oak at base age 50 years. Hardwood plot OH 2-B7 (age 71 yrs) was used in the analyses but is not depicted on the right-hand graph for consistency between the two graphs.

As expected, within the range of the mined hardwood data, the correlation between stand age and the tree ($r = 0.6211$, $P = 0.0007$) and ecosystem carbon pools ($r = 0.5675$, $P = 0.0025$) was positive (Table 1-4-4). This finding corroborates the results from the regression analysis for hardwood stands, for which the ecosystem carbon content increased asymptotically as stands aged across the range of SI (Figure 1-4-6). Sigmoid curves with asymptotic effect (“leveling off”) of biomass yield with increasing stand age are widely-used tree growth models for even-aged stands (Avery and Burkhart 1994b).

Conversely, despite the significant correlation between SI and ecosystem carbon for the hardwood stands (Table 1-4-4), the ecosystem carbon response surface for these stands suggested little or no effect of SI across stand age ranging between 25 and 60 years (Fig. 1-4-6). Evident from the regression equation depicted in Figure 1-4-6, the data for the hardwood stands suggested that SI had some effect on the amount of ecosystem carbon, but this effect was not statistically significant at the 90% probability level. For these stands, age was the sole significant parameter that could be used to model ecosystem carbon in mined hardwood forests (Fig. 1-4-6). Stand age successfully explained 36% of the variation in ecosystem carbon, and the remaining 64% was not accounted for in this analysis. In later sections of this paper, we present results of the regression analyses regressing projected carbon content at rotation age 60 years in each individual ecosystem component, tree, litter layer, and soil, by SI (independent variable); this was done to further study the zero effect of SI on the ecosystem carbon content in mined hardwood stands, which seemed counter-intuitive at first.

For mined mixed stands, the tree ($r = 0.5264$, $P = 0.0962$), ecosystem ($r = 0.5928$, $P = 0.0546$), and soil ($r = 0.8347$, $P = 0.0014$) carbon pools were positively correlated with site index, while litter layer carbon ($r = -0.7124$, $P = 0.0139$) was negatively correlated with SI (Table 1-4-4). As expected, the tree, soil, and total ecosystem carbon pools would increase with the increase of site index, based on the results from the correlation analysis. Similar to our discussion above for mined hardwood stands, the negative correlation of litter layer and SI under mixed vegetation might be explained by the influence of the hardwood component in these stands.

Although age was not significantly correlated with any of the mixed stands’ carbon pools (Table 1-4-4), the ratio of SI/Age was positively correlated with the tree, soil, and ecosystem carbon pools (Table 1-4-4). The graphical representation of the ecosystem carbon response surface by SI and age is presented in Figure 1-4-6. Since the majority of the mixed stands were dominated by pine trees (the hardwood tree basal area ranged between 20% and 40% of total plot BA), it was expected that the ecosystem carbon response surface for these stands would resemble that for pine stands (Figs. 1-4-5 and 1-4-6). The negative effect of stand age on ecosystem carbon content for mixed stands older than 25 years was in agreement with the results from the pine stand data analysis (Fig. 1-4-5). Similarly, ecosystem carbon content for mixed stands increased exponentially with the increase of SI.

Empirical Models for Total Tree, Litter Layer, and Soil Carbon Sequestration at Rotation Age 60 Years for Coniferous and Deciduous Forests on Mined Land. Comparison of sequestered Carbon on Mined and Non-Mined Land in the Eastern U.S.

Using species-specific growth rates, we projected tree biomass to a hardwood rotation age of 60 years for all study sites and developed ecosystem, tree, litter layer, and soil carbon regression models by site quality for the three forest types (Fig. 1-4-7). We also compared mined and non-mined carbon content models at rotation age by ecosystem component in hardwood stands. (Fig. 1-4-8).

We developed statistically significant and biologically reasonable models for ecosystem carbon content (Eco_C), describing the carbon levels expected at the end of a 60-year period, for the pine and mixed stands (Fig. 1-4-7). These carbon models and the model parameters were statistically significant ($P < 0.10$). In order to satisfy the assumptions for a regression analysis, namely, the normality and equal variance assumptions, we transformed (Montgomery et al. 2001) both the dependent (carbon estimates) and independent (SI) parameters for most of the analyses. For example, if Eco_C was transformed using the natural logarithmic function, $\text{Eco\_C}' = \ln(\text{Eco\_C})$, and the regression model was of the form $\text{Eco\_C}' = f(\text{SI})$, then the regression model for Eco_C would be written as $\text{Eco\_C} = \text{Exp}[f(\text{SI})]$, where Exp is the exponential function. We used the non-transformed carbon data if they were normally distributed and the variance was equal across the spectrum of analysis.

Across the range of mine site quality, SI ranging from 10 to 35 m, the mixed and pine stand models explained 34% and 52%, respectively, of the variation in Eco_C. For both regression models, an increase in site index resulted in an increase in Eco_C, so that for each 1-m increase in site index, approximately 5 and 13 Mg C ha⁻¹ more Eco_C were sequestered during a 60-year period under mixed and pine vegetation, respectively (Fig. 1-4-7).

These results indicated that if low-quality mined sites planted to mixed and pine stands were restored to a site quality comparable to the average of non-mined sites (SI = 25.5 m), then the SI of the sites in this study would be increased up to 10.5 m and 6 m for mixed and pine stands, respectively. The increase in SI would result in 55% and 74% increase of the Eco_C sequestered on mined sites planted to mixed (pine and hardwood) and pine vegetation, respectively, for a 60-year carbon sequestration project (Fig. 1-4-7).

Ecosystem carbon increased exponentially with increase in SI in pine and mixed stands. Ecosystem carbon content in mined hardwood stands was not affected by changes in SI (Fig. 1-4-7), which was also depicted in Figure 1-4-6. With an average estimate of 153 Mg C ha⁻¹, the 60-year estimate of Eco_C in hardwood stands was significantly lower than the estimates for pine stands on high quality sites, SI ≥ 28 m (Figure 1-4-7). This was a logical outcome as the pine stands were modeled assuming stand management at a 30-year rotation that would lead to higher levels of sequestered carbon after 60 years; that is, biomass and carbon associated with two 30-yr rotations compared to one 60-yr rotation for the hardwood stands. Although the Eco_C estimates in mixed stands were greater than those in hardwood stands on high quality sites, these differences were not significant, most likely due to the hardwood component in the mixed stands. Similarly, due to the pine component in the mixed stands, the differences of Eco_C estimates in pine and mixed stands were not significant across the spectrum of site quality.

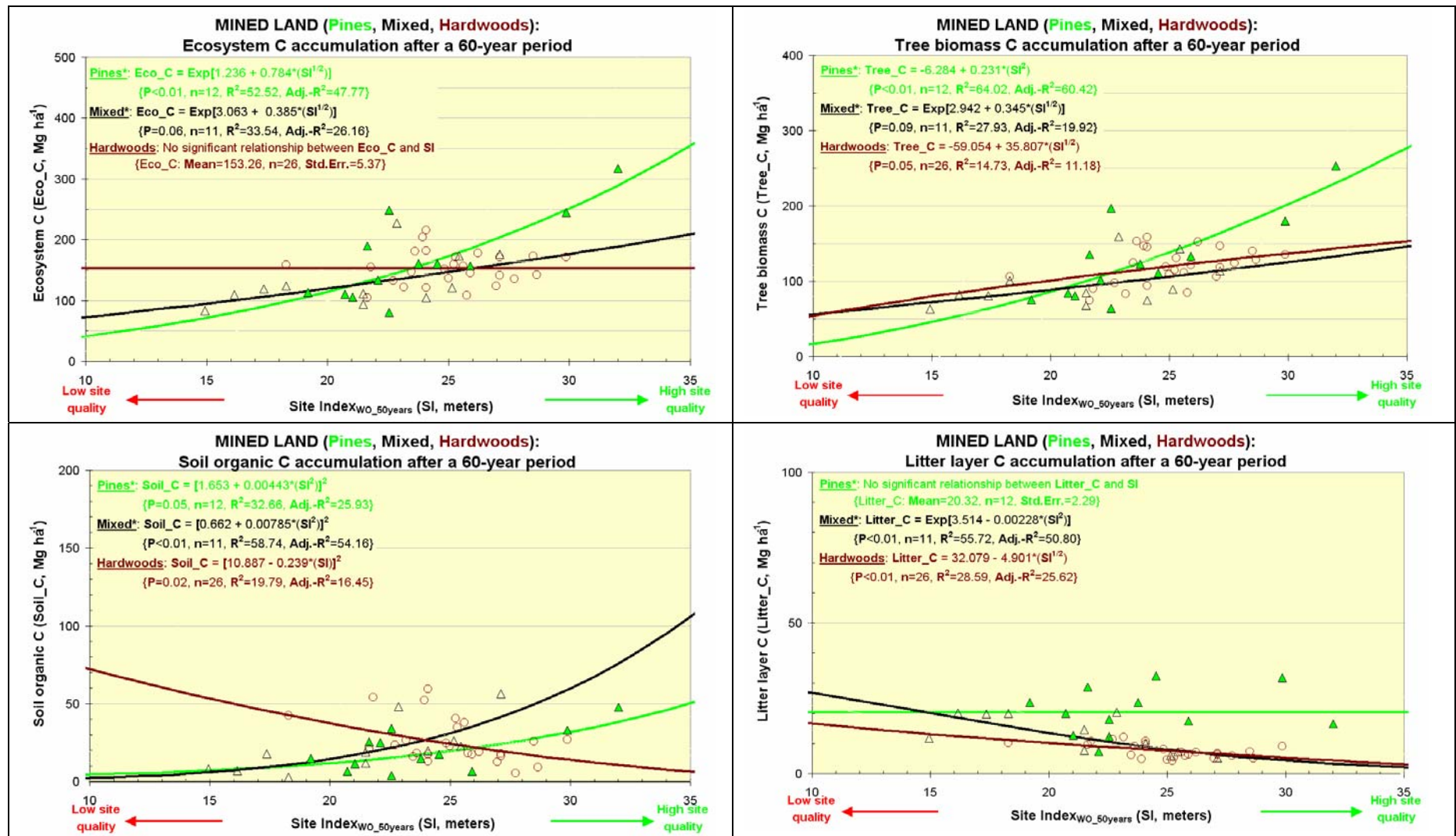


Figure 1-4-7. Regression models for total ecosystem, tree, soil, and litter layer carbon content projected to rotation age 60 years on forested mined land for three forest types, pine, mixed, hardwood, in the Midwestern and Appalachian coalfields. NOTE: models followed by * were developed without data outliers, as described in the text. For consistency among the graphs, the horizontal grid lines indicate 50 Mg C ha⁻¹ increments. Site index is the height for white oak at base age 50 years.

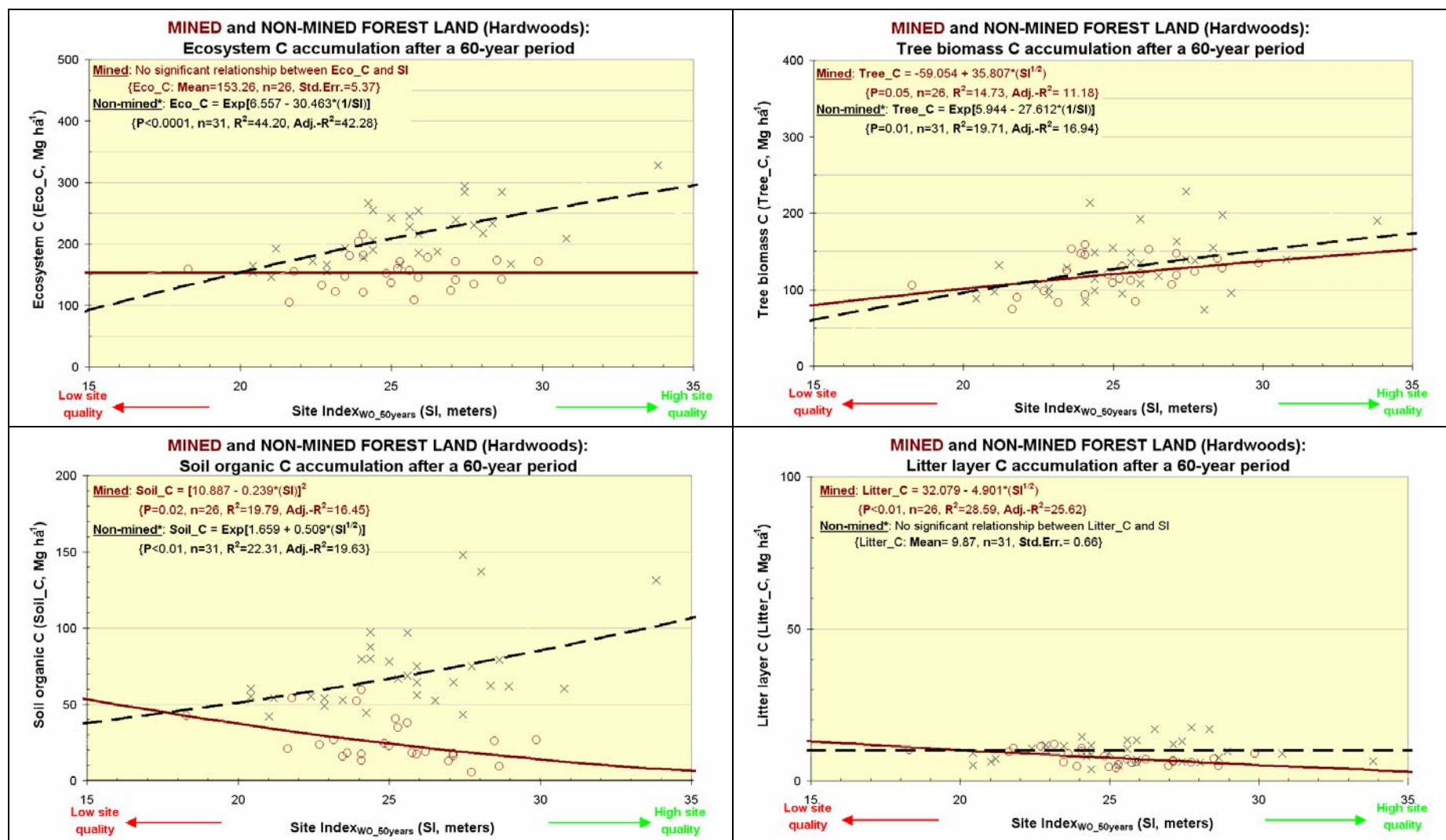


Figure 1-4-8. Comparison of regression models for total ecosystem, tree, soil, and litter layer carbon content projected to rotation age 60 years in hardwood stands on mined and non-mined land in the Midwestern and Appalachian coalfields. NOTE: models followed by * were developed without data outliers, as described in the text. For consistency among the graphs, the horizontal grid lines indicate 50 Mg C ha⁻¹ increments. Site index is the height for white oak at base age 50 years.

Although the results of the hardwood Eco_C model were not as expected, the negligible cumulative effect of SI on Eco_C appeared reasonable after analyzing the individual tree, litter, and soil carbon pools for the hardwood stands. The tree carbon (Tree_C) regression model for the hardwood stands indicated that there was a slight positive effect of SI on the amount of carbon sequestered and stored in tree biomass. However, hardwood litter and soil carbon content decreased with site quality (Fig. 1-4-7). At the current young stage of mine soil development, soil conditions on high-quality sites are more favorable for a myriad of soil micro- and macro-organisms (Showalter et al. 2007), which decompose the organic matter in the litter layer and soil to CO₂ at higher rates. As a result, the cumulative effect of site quality on the ecosystem carbon content under hardwood vegetation was negligible (Fig. 1-4-7). As mined stands age, we expect higher quantities of soil carbon under hardwood vegetation to become protected from microbial decomposition, via clay particle aggregation (Puget et al. 2000) and Al-, Fe-chelation of soil organic matter (Masiello et al. 2004), thus resulting in higher SOC stock as SI increases toward the SOC levels in non-mined forests (Fig. 1-4-7).

Similar to Eco_C, the differences in Tree_C in hardwood and mixed stands were not significant. Despite the steeper model curve depicting Tree_C accumulation in pine stands in Fig. 1-4-7, these estimates were not significantly different from Tree_C in hardwood and mixed stands across the spectrum of SI. This could be a result of the high variability of the Tree_C estimates and the relatively young age of the mined forest stands. Increased variability of tree carbon could mask the effect of site quality for different forest types. As stands age and the mine soils develop and become more homogenous, we expect that Tree_C in more intensively managed forests, such as pine and mixed stands managed at 30 year rotations, would sequester significantly more carbon in tree biomass and wood products manufactured from harvested wood from these forests. We chose a 30-year rotation length for the mixed stands because the yellow pine species are commonly managed at that interval, and because we lack stand data to predict pine tree mortality for a 60-yr rotation.

In Figure 1-4-9, we compared the ecosystem carbon model for mined mixed stands under two management scenarios, two 30-year rotations (2x30yr) and one 60-year rotation (1x60yr). The model showing ecosystem carbon content after two 30-year rotations was the same as depicted in Figure 1-4-7. The ecosystem carbon estimates for the 60-year rotation scenario for mined mixed stands were the same as depicted in Figure 1-4-6. The 3-D response surface in Figure 1-4-6 was projected to a two-dimensional plane for which age was equal to 60 years. The results from this comparison showed that for medium-low to high-quality sites (SI > 12m) the Eco_C estimates for the 2 x 30yr scenario were higher compared to the 1 x 60yr scenario. The differences in Eco_C between the two management scenarios were significant for medium-quality sites ranging from 22 to 30 m of SI; the average SI of the mixed stands in this study was 21 m. For medium-quality sites, the Eco_C estimates for the 2 x 30yr scenario were approximately 53%, ranging from 41% (SI = 22 m) to 66% (SI = 35 m), higher than the Eco_C results for the 1 x 60yr scenario. In general, for each 1-m increase in site index, approximately 5 and 2 Mg C ha⁻¹ more Eco_C were sequestered during a 60-year period under the 2 x 30yr and 1 x 60yr management scenarios, respectively (Figure 1-4-9).

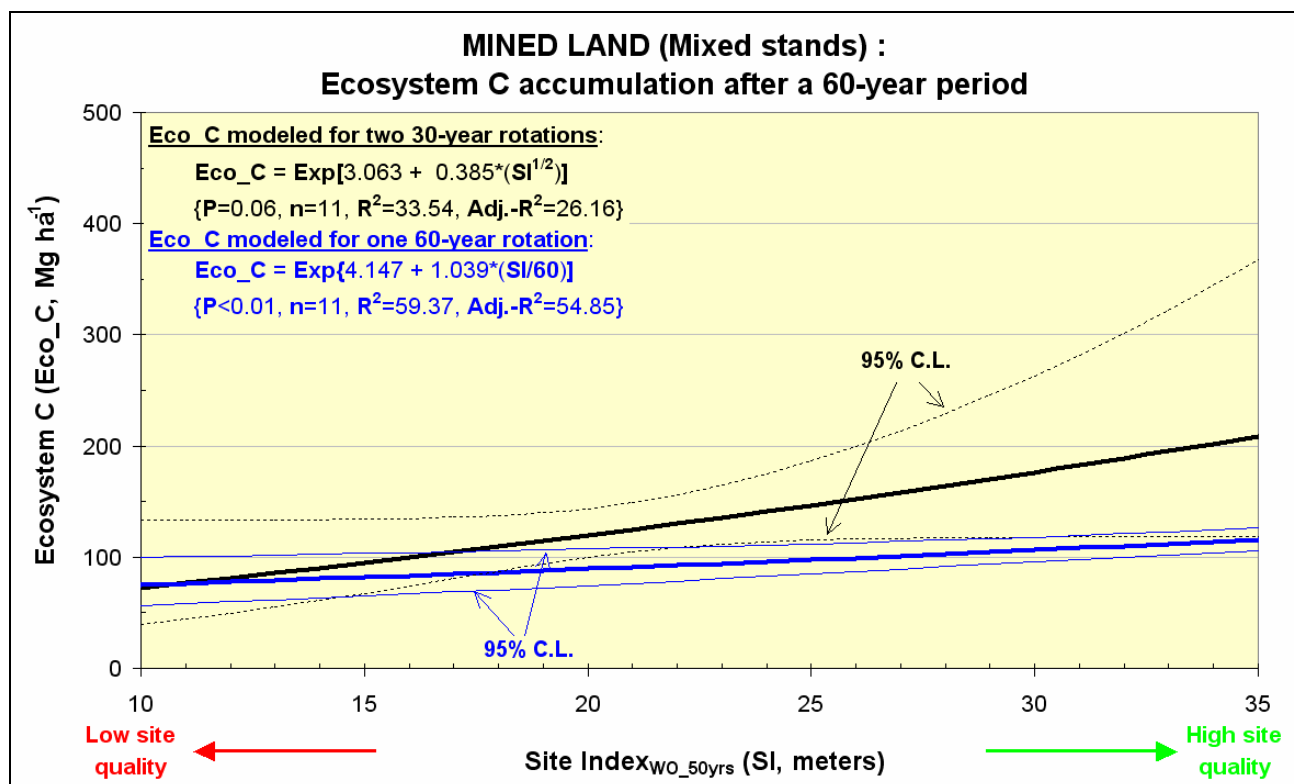


Figure 1-4-9. Comparison of regression models for total ecosystem carbon content projected to rotation age 60 years for two forest management scenarios, two 30-year rotations and one 60-year rotation, for mined mixed stands in the Midwestern and Appalachian coalfields. Site index is the height, in meters, for white oak at base age 50 years. Dashed lines indicate the 95% confidence limits (C.L.) of the respective model mean estimates.

Based on these results, we believe the 30-year rotation for mixed stands would be the better management scenario when maximizing carbon sequestration. These results should be used carefully as they include no data analysis about the market value of saw timber (for 1 x 60yr scenario) and pulpwood (for 2 x 30yr scenario), nor any monetary estimates depicting carbon credit for the carbon sequestered in forest biomass, nor any data describing the carbon credits from a substitution effect of pulpwood usage for energy production, instead of burning fossil fuels for energy. A more complete analysis including the abovementioned data may produce different results and could suggest different management scenarios for mined mixed stands.

The regression analysis of the projected litter layer carbon (Litter_C) data in all stands (Fig. 1-4-7) supported our earlier findings from the correlation analysis presented in Table 1-4-4; there were no significant changes in the litter carbon pool with increasing SI under pine vegetation and the pool decreased with SI under hardwood and mixed forest vegetation (Fig. 1-4-7). The average Litter_C in the pine stands, 20 Mg C ha⁻¹, was significantly higher on medium- to high-quality sites (SI > 22 m) compared to mixed stands. The pine stands' litter carbon pool was significantly higher than Litter_C under hardwood vegetation for mined lands of medium-low, medium, and high site quality, SI > 17m. These results were expected because pine litter decomposes slower and accumulates at faster rates than hardwood litter.

Similar to the tree carbon pool, the differences between Litter_C in mixed and hardwood stands were not significant across the spectrum of SI. This similarity could be due to the hardwood component in the mixed stands. There was a decrease of both the hardwood and mixed stands' Litter_C as site quality increased (Fig. 1-4-7). Similar to our discussion about Eco_C, above, we believe that better site conditions favor higher microbial activity within the litter biomass, resulting in faster litter decomposition at the current stage of soil and site development on the hardwood and mixed forest stands.

Soil organic carbon increased with increasing SI in mixed and pine stands, but the opposite occurred in hardwood stands (Figure 1-4-7). For each 1-m increase in site index, approximately 4 and 2 Mg C ha⁻¹ more SOC were sequestered during a 60-year period under mixed and pine vegetation, respectively (Figure 1-4-7). The effect of SI on SOC accumulation was greatest in mixed stands, most likely due to the combined effect of both pine and hardwood vegetation. On high-quality sites, as indicated by the mixed stand Tree_C model (Fig. 1-4-7), higher tree biomass would result in higher root biomass and litter inputs, the combined effect of which would be higher organic matter input into the soil. We believe that, due to the specific site conditions created under mixed vegetation, mainly due to the pine component, i.e., relatively acidic soil and litter layer environment, lower nutrient availability, and high-lignin, high-wax chemical composition of the foliage under mixed vegetation, SOC decomposition rates were lower in mixed stands compared to hardwood stands. Therefore, on high-quality sites, more SOC would accumulate in mixed stands than pure hardwood stands.

Our regression models indicated that SOC in mixed stands was significantly higher than SOC in hardwood stands on high quality sites, SI > 28 m, but was significantly lower on medium and low quality sites, SI < 20 m (P < 0.05). On medium- and low-quality sites, hardwood SOC was also significantly higher than pine SOC. Interestingly, our models suggested that the mixed stand SOC estimates were not significantly different from those of the pine stands across the entire spectrum of SI (P < 0.05) (Fig. 1-4-7). The latter could be due to the effect of the considerable pine component in the mixed stands.

Based on the ecosystem carbon models for the pine, mixed, and hardwood stands on the mined sites (Fig. 1-4-7), it is clear that pine stands managed on 30-yr rotations could sequester more carbon than mixed or hardwood stands if mined sites are restored to higher quality. Alternatively, mixed stands may provide additional ecosystem services such as wildlife habitat and aesthetic value for coalfield communities. Our models show that establishing pine forests or mixed forests with pine and hardwood tree species compared to pure hardwood forest stands would sequester approximately 85% more carbon and would respond better to site quality improvement practices.

The Eco_C model for non-mined hardwood stands accounted for 44% of the variation in ecosystem carbon as a function of SI. Ecosystem carbon in the non-mined hardwood stands increased with the increase of SI (Fig. 1-4-8). Non-mined Eco_C was significantly greater than mined Eco_C for high quality sites, SI > 22 m, and was similar for low-quality sites. This corroborates the observations of other researchers who reported that post-mining productivity can be greater on originally poor sites (SI < 20 m), but that post-mining productivity is less on medium- to high-quality sites (Ashby et al. 1980; Burger et al. 2003).

The non-mined Litter_C content was not affected by changes in SI, averaging 10 Mg C ha⁻¹. With the exception of high-quality sites (SI > 26 m), for which the litter carbon pool in non-

mined stands was significantly higher compared to mined hardwood stands, the Tree_C and Litter_C regression models showed that mined and non-mined hardwood forests sequestered similar amounts of carbon in tree and litter layer biomass as a function of SI (Fig. 1-4-8). Therefore, increasing Eco_C with increasing SI on non-mined sites was due to the greater amount of sequestered SOC component in non-mined hardwood stands compared to mined hardwood stands.

On medium- to high-quality sites ($SI > 21$ m), non-mined hardwood stands sequestered significantly higher SOC than mined hardwood stands. In general, during a 60-year period, for each 1-m increase in site index, approximately 3 Mg C ha^{-1} more SOC was sequestered in non-mined hardwood stands, while 2 Mg C ha^{-1} of soil carbon was lost as CO_2 in the mined hardwood stands (Fig. 1-4-8).

It should be noted that using the coal-correction equations from Substudy 1 to correct the Walkley-Black carbon values of older mine soils might result in a certain over-correction for the coal error. Samples from young mine soils were used to develop the coal correction equation described above (Substudy 1). The iron present in young mine soils is predominantly in its reduced form, ferrous iron (Fe^{2+}), which, similar to coal, is the reason for soil carbon overestimation when the Walkley-Black procedure is used for soil analysis (Substudy 1). However, as the overburden spoil material ages and weathers, such as the older mined sites characterized by Rodrigue (2001), the ferrous iron is oxidized to ferric iron (Fe^{3+}), which does not interfere with the Walkley-Black procedure.

It could be hypothesized that the overburden material of mined sites of higher site quality is relatively more weathered than lower quality sites; hence, the ratio $\text{Fe}^{2+}/\text{Fe}^{3+}$ would decrease (i.e., less Fe^{2+} is present) with the increase of site quality. If the above were found to be true, then applying the coal-correction equation from Substudy 1 to correct Walkley-Black carbon would result in higher over-correction of values for mined sites of high site quality compared to low-quality mined sites. Such relationship between SI and spoil weathering would explain the decrease in SOC content with the increase in SI in mined hardwood stands, which was not as expected (Fig. 1-4-7). However, the above hypothesis was not supported by the SOC data for pine and mixed stands, which were similar (in age and site quality) to the hardwood stands. The results for pine and mixed stands indicated that SOC content increased with the increase of SI, which was as expected (Fig. 1-4-7). It is logical to assume that the over-correction of the Walkley-Black carbon values reported by Rodrigue (2001) would be consistent across SI for all mined sites, without any discrimination of vegetation type. Therefore, the decreasing trend of SOC with the increase in SI in mined hardwood stands is most likely a result of the combined effects of many biotic and abiotic factors, as opposed to being an artifact of the coal-correction procedure used in this study.

The results of the comparison of the carbon pools between mined and non-mined hardwood stands in Figure 1-4-8 showed that Tree_C and Litter_C pools were relatively unaffected by the mining practices and that hardwood tree growth and litter accumulation rates were restored to the pre-mining levels. Over time, the soil carbon pool will increase and will eventually attain a new equilibrium level, probably similar to that of soils in the non-mined sites. This is the empty cup representing the important distinction between mined and non-mined land. The amount of SOC that could be sequestered along this site quality gradient ($SI > 22$ m) ranges from 25 to 100 Mg C ha^{-1} . Although site quality appears to have little effect on aboveground carbon sequestration in hardwood vegetation on mined compared to non-mined sites, the non-mined SOC regression

model shows that site quality has a very significant effect on SOC potential. High post-mining site quality creates a larger empty cup to be filled with SOC produced with either pine or hardwood forest stands.

The higher the original forest site quality, the less likely productivity was restored, and the greater the disparity between pre- and post-mining ecosystem carbon sequestration potential. Despite the observed similarities of tree growth and litter biomass accumulation before and after mining, surface coal mining could decrease the carbon sequestration potential of mined sites by interrupting the biogeochemical cycle of terrestrial carbon. On natural sites, the carbon sequestered by live vegetation and deposited above ground as leaf litter and dead organic matter is decomposed by soil microorganisms and, ultimately, a fraction of this organic matter is incorporated in the soils. However, during coal mining (earth removal, transportation, storage, mixing), microbial communities in the soil could be completely destroyed, thus, interrupting the natural cycle of terrestrial carbon. Therefore, a strong argument could be made that coal operators use reclamation practices that restore forests to their pre-mining carbon sequestration potential. In particular, better site reclamation methods should focus on introducing, accumulation, and stabilization of soil organic matter in mine soils, if soil microbial communities are to be re-established, and the long-term carbon sequestration potential is to be restored and maintained after coal mining.

Conclusions

We characterized 14 mined and eight adjacent non-mined forests throughout the Midwestern and Eastern coal regions to determine the carbon sequestration potential of mined land reclaimed prior to the passage of the Surface Mining Control and Reclamation Act of 1977. On average, the highest amount of ecosystem carbon on mined land was sequestered by pine stands (148 Mg ha^{-1}), followed by hardwood stands (130 Mg ha^{-1}) and mixed stands (118 Mg ha^{-1}). Non-mined hardwood stands sequestered approximately 42%, 62%, and 79% more cumulative carbon in total tree biomass, litter, and soils, than the mined pine, mined hardwood, and mined mixed stands, respectively. For the mined forest stands, approximately three-quarters of the total ecosystem carbon was sequestered in the total tree biomass (stem wood, stem bark, foliage, treetops, branches, stumps, and coarse roots) and the remaining one-quarter was distributed between the litter and soil carbon pools. Site quality had an exponential effect on ecosystem carbon content in both pine and mixed stands, and these models showed a decrease in ecosystem carbon with stand age due to natural pine tree deterioration occurring in older pine stands and in the pine component of mixed stands. Ecosystem carbon increased asymptotically with stand age in the mined hardwood stands but was not affected by site quality.

The development of productive forests on reclaimed land satisfies multiple goals. Successful reforestation on productive mined land meets SMCRA guidelines that require the return or enhancement of pre-mining productivity levels. Reforestation also establishes a long-term sink for atmospheric carbon. Considering that very little organic carbon is present on a recently reclaimed mined site, there is great potential for sequestering carbon by restoring the forest to a level of productivity equal to or greater than that present before mining.

The tree and litter carbon pools were similar on mined and non-mined hardwood stands across the entire range of site quality. However, mined land SOC levels were lower than average natural sites due to the pre-law mining process, which disrupts the biogeochemical cycle of all elements, including C. By removing the forest and soil fauna and flora, the carbon cycle is

breached and sequestration is interrupted. Reincorporating organic matter into the soil requires establishment of a biological community, including plants for primary production, and soil fauna and flora to process the detritus.

Surface coal mining procedures used in the Midwestern and the Appalachian coal fields of the U.S. prior to 1977 could destroy the microbial communities in the soils that existed prior to mining. Some of the implications of eliminating the soil fauna and flora from the original soil due to land disturbance could lead to a drastic reduction of the potential of mined forest ecosystems to sequester carbon at pre-mining levels. Based on the carbon sequestration models generated in this study at rotation age 60 years, the non-mined hardwood stands sequestered significantly more soil carbon and, as a result, significantly more total ecosystem carbon on high quality sites ($SI > 22$ m) compared to similar hardwood stands established on mined lands. On low-quality sites, the ecosystem carbon levels were similar for mined and non-mined forest stands.

Another important outcome from this study was a set of empirical models developed to describe terrestrial carbon sequestration in total tree biomass, litter layer, and soils for pine, mixed, and hardwood forests established on mined land in the Midwestern and Appalachian coal fields. These equations could be used as building components of a decision support system (DSS) for predicting the amount of sequestered carbon in tree biomass, litter, and the soil for different forest types and different forest establishment practices designed for mined land. The greater good of a DSS would be the ability to make informed decisions about what type of forests to establish on mined lands that would maximize the carbon sequestration potential of the land and that would be most beneficial to mined land owners, carbon sequestration scientists, and society.

Task 1, Sub-study 5: A Decision Support System for Estimating Carbon Sequestered on Mined Land

Experimental

The goal of this research was to develop a decision support system (DSS) that would allow prediction of the amount of sequestered carbon within each phase of the carbon cycle, including the live forest vegetation biomass, root biomass, soil carbon pool, and biomass that has been harvested and transformed into wood products, such as house-building materials, furniture, and paper products. We used published data and knowledge to define the following components of the proposed DSS: (i) hybrid poplar ecosystem carbon models, (ii) pine and hardwood stand growth models, (iii) decomposition rates of leaf litter and roots for poplar, pine, and hardwood stands, (iv) effects of weed control, tillage, and fertilization on the survival and growth of trees planted on reclaimed mined lands, (v) effects of forest management on changes of the soil organic carbon (SOC) pool, and (vi) carbon storage in wood products in use, in landfills, and carbon emissions from biomass burning and fossil fuel usage during forestry and wood product manufacturing operations.

Hybrid Poplar Growth Models

Diameter Growth by Stand Age. Coleman et al. (2000) reported an average diameter growth of 0.54 cm yr^{-1} over a 3-year period for hybrid poplar plantations (*Populus tristis* x *P. balsamifera*) near Rhinelander, WI, established on well-drained Padus sandy-loam soils underlain by glacial outwash and stratified sand. The mean diameter of these 3-year-old trees was 13.0 cm and their stocking was at 1619 trees ha^{-1} (Coleman et al. 2000).

Another group of researchers investigating hybrid poplar-growth reported the detection of quantitative trait loci (QTLs) responsible for growth trajectories in hybrid poplar trees, using the *P. deltoides* x *P. euroamericana* (*P. deltoides* x *P. nigra*) hybrid (Wu et al. 2003). A total of 450 one-year-old rooted hybrid seedlings were planted at 4 x 5 m spacing at a forest farm in the Jiangsu province in China; height and diameter were measured at the end of every growing season, for a total of 11 seasons (Wu et al. 2003). Based on standard growth-model theories, Wu et al. (2003) developed a novel statistical method to depict poplar tree growth as a continuous function with respect to time. Their method used the universal S-shaped curve model, which is commonly used in biology (Wu et al. 2003).

The poplar growth curve reported by Wu et al. (2003) agreed adequately with the growth of hybrid poplar trees reported by Roseberg (2004). The latter researcher reported growth data for hybrid 52-225, *P. trichocarpa* x *P. deltoides*, which was grown on sandy loam soil at the Southern Oregon Research and Extension Center Hanley Farm for eight growing seasons (Roseberg 2004). The stand was irrigated with standard irrigation water and was also fertilized each year in early summer with 58-141 kg ha^{-1} N and one treatment of 52 kg ha^{-1} of P_2O_5 in the fifth year. The diameter at breast height (DBH) of 2-year-old hybrid poplar trees averaged 8.4 cm, and was 16.5 and 20.2 cm at stand ages 5 and 8 years, respectively (derived from Roseberg 2004).

Tree Biomass Estimation Using DBH. Ney et al. (2005) measured 1478 trees on approximately 0.4 ha in an existing 7-year-old stream buffer of hybrid poplar trees planted near Amana, Iowa. They selected 35 random trees, which were removed, and total height was determined with branches and leaves removed. The authors reported a highly accurate

regression model (R-square of 99.22%) for prediction of the total tree wet weight (W, pounds) from measured DBH (cm) of hybrid poplar trees, $\ln(W) = -3.6143 + 2.2686 \ln(DBH)$. The biomass moisture content was measured, 46.56-47.91%, at 95% confidence level, and was used to convert wet weight to dry weight before estimating carbon stocks (Ney et al. 2005). Results showed that carbon storage in above-ground biomass accumulated at a rate ranging from 0.92 to 1.37 Mg C ha⁻¹yr⁻¹. The carbon stock in above-ground biomass for this 7-year-old stand was estimated to be 3.57 to 3.71 Mg ha⁻¹ (Ney et al., 2005).

Similarly, Netzer et al. (2002) collected DBH (cm) data from repeated measures on selected hybrid poplar trees, and cut, oven-dried trees for dry stem weight measures in kg units, to develop biomass equations of the following general form, $Tree\ weight = a + b*DBH + c*DBH^2$. The authors surveyed poplar plantations established in 1987-88 in Minnesota, Wisconsin, North Dakota, and South Dakota on previously agricultural land that was managed for crops, small grain, or hay the year before hybrid poplar stand establishment. The plantations surveyed ranged from 7 to 12 years old, all planted at 2.4 x 2.4 m spacing, resulting in 1736 trees ha⁻¹. Soil textures ranged from loamy sand to silty clay loam. Before planting, fall weed control was applied to remove perennial grasses, and deep plowing and disking were also used. After analysis of the data for all hybrid clones, the authors reported a dry stem biomass prediction equation of relatively high R-square value (R²=98.30%, sample size equal to 152), $Tree\ weight = 6.16 - 2.23*(DBH) + 0.353*(DBH)^2$. The mean annual DBH increase on all plantations ranged from 1.34 to 2.39 cm yr⁻¹, with average of 1.778 cm yr⁻¹ (Netzer et al., 2002).

Site Quality Effect on Hybrid Poplar Biomass Yield. A hybrid poplar production study was established in the Willamette Valley, Oregon, as part of a study project at Oregon State University (Hibbs and Bower, 1998). Height, diameter, age, and density measurements collected from 160 plots were analyzed to predict yield. The authors used soil quality, similar to site index, to differentiate between different yield classes. They reported the biomass volume yield of hybrid poplar between ages 1 to 10 years for four soil productivity classes, high, medium-high, medium-low, and low. On average, the biomass volume on high quality sites, site index (SI) of 33 m or greater, was approximately twice that observed on low quality sites, SI of 23 m (derived from Hibbs and Bower, 1998). Their data also suggested that the biomass yield of hybrid poplar decreased approximately 5.8% for every 1-m decrease in SI (derived from Hibbs and Bower, 1998). The decrease of biomass yield due to reduction in soil quality ranged from 6.7% to 5.0% per 1-m decrease in SI across stand age (between 3^{1/2} and 9 years) (Hibbs and Bower, 1998).

Biomass Distribution to Leaves, Branches, Stems, and Roots. In a field experiment in Washington, Wullschleger et al. (2005) reported the relative biomass distribution for each tree for 1- and 2-year-old clones to stems, branches, leaves, coarse roots, and fine roots as a fraction of total dry mass. Two hybrid poplar families of the *Populus trichocarpa* x *Populus deltoides* clone were planted on active dune soils and were irrigated and fertilized (~80 kg ha⁻¹ yr⁻¹ N). Allometric equations were developed to predict the biomass in each tree component using a regression equation of the $Y=b*X^a$ general form, derived from $\log(Y)=\log(b)+a*\log(X)$ equation, where X and Y are independent variables of plant biomass, measured in gram units (Wullschleger et al., 2005). Among other results, allometric equations to predict the biomass content in tree leaves, stem, branches, and roots for 2-year-old hybrid poplar trees were reported (Wullschleger et al., 2005): $leave\ biomass\ (g/tree) = 0.3167 * (TotBIOM, g/tree)^{0.9621}$, R²=0.966; $stem\ biomass, g/tree = 0.2787 * (TotBIOM, g/tree)^{1.0298}$, R²=0.910; $branch\ biomass, g/tree =$

$0.0852 * (TotBIOM, \text{g/tree})^{1.1004}$, $R^2=0.890$; root biomass, $\text{g/tree}= 0.4315 * (TotBIOM, \text{g/tree})^{0.9057}$, $R^2= 0.903$, where *TotBIOM* (*g/tree*) is the total tree biomass (Wullschleger et al., 2005).

Wullschleger et al. (2005) reported that, in general, 62% (+/- 9 %) of total tree biomass is distributed to shoots, and 38% (+/- 9 %) to roots during the establishment year, and 79% (+/- 4%) to shoots and 21% (+/- 4 %) to roots in 2-year-old trees. The authors determined that as hybrid poplar stands developed, the biomass was preferentially allocated to stems and branches, when compared to root biomass. Significant effects of genotype on biomass distribution were also observed (Wullschleger et al. 2005).

Leaf Litter and Root Biomass Decomposition Rates. In a study by Cotrufo et al. (2005), established in central Italy, second and third growing season litter production were reported for three *Populus* species, *P. euroamericana*, *P. nigra*, and *P. alba*. Litter was collected twice a month throughout the growing season and decay rates were computed by tree species. Mean annual litter accumulation across the three species was $5.34 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ and $6.40 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ for 2nd and 3rd seasons, respectively. Decay models were successfully extrapolated beyond the 8-month measurement period to produce litter decomposition models (Cotrufo et al. 2005). Decay constants, k (yr^{-1}) for the three species were 1.12 yr^{-1} (*P. nigra*), 2.01 yr^{-1} (*P. euroamericana*), and 2.86 yr^{-1} (*P. alba*) (Cotrufo et al. 2005), indicating that the time for approximately 95% of the litter to be mineralized was 2.68, 1.49, and 1.05 years for the respective species.

Coleman et al. (2000) reported root dynamics data for hybrid poplar plantations (*P. tristis* x *P. balsamifera*) near Rhinelander, WI. Root biomass analysis indicated that approximately 46% of the total root biomass was present as fine-roots (< 0.5 mm diameter) and 54% as coarse roots (> 0.5 mm diameter), where the total root biomass was estimated at 768 g m^{-2} with an estimated standard deviation of 161 g m^{-2} (Coleman et al. 2000). The authors also reported that almost all roots (94%) disappeared after 700 days, compared to 36% for red pine roots analyzed in the same study. Seasonal effects were significant on fine- and coarse-root production and mortality, with a majority appearing between July and September (Coleman et al. 2000).

SOC Accumulation. Soil organic carbon accumulation under hybrid poplar stands was reported by Hansen (1993). Eleven hybrid poplar plantations were located across North Dakota, Minnesota, and Wisconsin, where the trees were at least 4 years old and the plantations were adjacent to crop/grassland, among other selection criteria (Hansen 1993). Trees were planted on previously tilled soils that were originally prairies. Soil samples were collected from 3 depth zones, 0 cm to plow pan (including litter layer), plow pan to 50 cm, 50-100 cm. Soil carbon and soil bulk density were measured; tree DBH was measured and tree dry weight were estimated using biomass regression equations from the literature (Hansen 1993). The results indicated that young plantations (4 to 6 years old) had less SOC than adjacent crops, indicating that SOC mineralization occurred due to plantation establishment, primarily within the 0-30 cm depth. Hansen (1993) developed a regression equation to predict the net SOC accumulation within the upper 100 cm of the soil profile in hybrid poplar plantations, in which the soil carbon difference [*Tree SOC (+ litter carbon) – crop SOC*] was expressed as a function of stand age.

Hansen (1993) reported that the net SOC accumulation, including the litter layer carbon, could be predicted by stand age using a regression equation of relatively high R-square ($R^2 = 76\%$): [*Tree SOC (+ litter carbon) – crop SOC*] = $-43.5 + 4.48 * \text{Age}$. The authors also reported that SOC (+ litter carbon) under hybrid poplar stands was 12 Mg ha^{-1} more than adjacent row crops, and 34 Mg ha^{-1} more than mowed grassland. The oldest plantation (age 30) in their study

had SOC of 96 Mg C ha⁻¹, indicating an accumulation rate of 3.2 Mg C ha⁻¹ yr⁻¹. It was suggested that plantations be grown on longer rotation lengths, greater than 4-6 years, to avoid the initial carbon loss (Hansen 1993).

In Substudy 2, we discussed the significance of determining a site-specific maximum cost-effective depth (MCD) for SOC analysis in mine soils, which is to be completed prior to the actual SOC inventory. In summary, MCD is the soil depth at which the costs of SOC measurement, including sample collection, preparation, and analysis, break even with the revenues received from claiming carbon credits for the SOC sequestered to the 0-MCD [cm] depth of the mine soil. Therefore, if SOC analysis across a mined land is desired for lower depths than MCD, then the costs may exceed the expected profits. Some of the factors that define the MCD of a certain site are the horizontal variability of SOC on the mined site, expressed as the number of soil pits required to measure soil carbon at accepted accuracy (e.g. 15% within the true mean) and precision levels (e.g. at 95% probability level), the current market price of C, and the relative costs of SOC analysis in mine soil, including costs of excavation, sample preparation, and laboratory sample analysis.

Because each mine soil has unique chemical and physical properties characterized by the parent spoil material, the vegetation, and the reclamation techniques used, it is nearly impossible to develop a universal equation that would predict the cost-effective, measurable fraction of SOC across a certain mined site. However, using site-specific data for determining both the cumulative distribution of SOC down the soil profile and the MCD estimate, one could reasonably predict the cost-effective, measurable fraction of sequestered SOC on mined land.

Growth Models for Pine and Hardwood Stands

Avery and Burkhart (1994b) discussed the components of growth and yield models that are used operationally in forestry. They summarized that the factors that most directly describe the growth (biomass increase over a period of time) and yield (total harvestable biomass at a given time per unit area) of forest stands are stand age, site index, and stand density. Yield curves are often used to predict current and future tree biomass production so that the most appropriate management regime could be selected to maximize total wood production for a certain tree species, site quality, and desired wood products. For a specific site quality and initial stand density, the curve describing yield by stand age approximates a sigmoid curve (Figure 1-5-1) (Avery and Burkhart, 1994b).

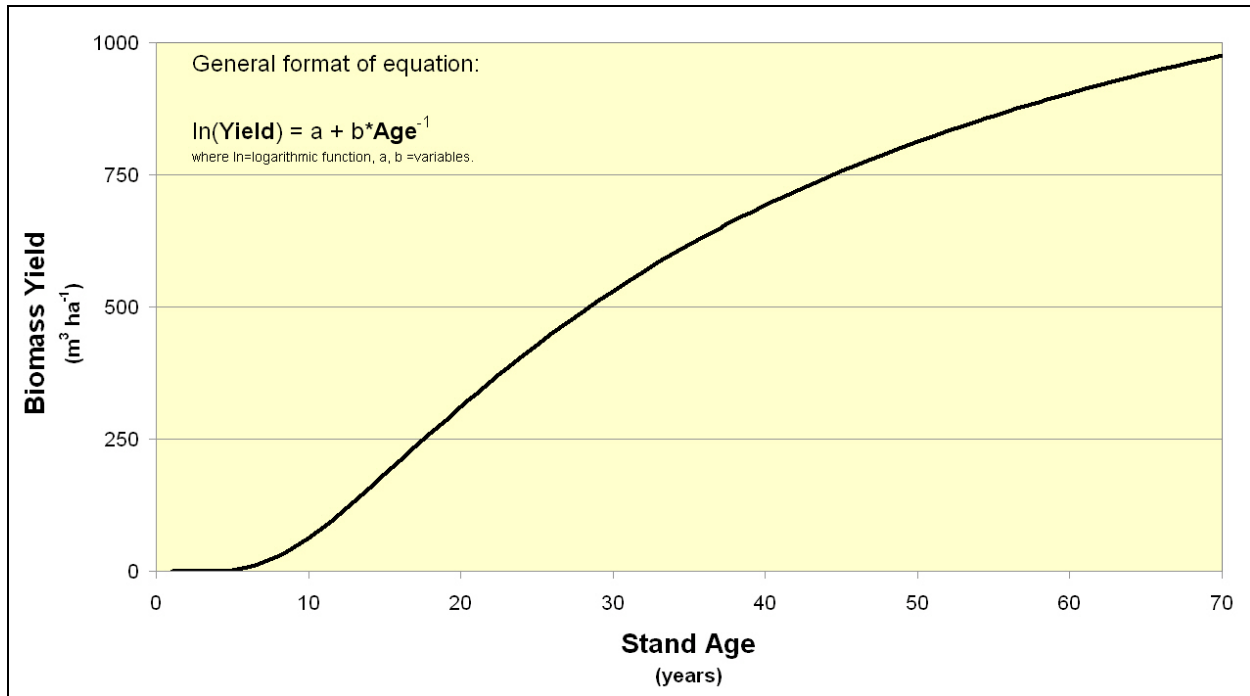


Figure 1-5-1. Biomass yield prediction by stand age for forest stands established at a given stand density and site index (adopted from Avery and Burkhart, 1994b).

When both site index and stand age data are used as indicator variables to predict tree biomass yield in a multiple regression analysis, the following commonly-used growth and yield model could be utilized for analyses (Brender 1960; Brender and Clutter 1970; Clutter 1963; Nelson et al. 1961):

$$\ln(\text{Yield}) = b_0 + b_1(SI) + b_2(\text{Age}) + b_3(SI^2) + b_4\left(\frac{1}{\text{Age}}\right) + b_5\left(\frac{1}{\text{Age}^2}\right) + b_6\left(\frac{SI}{\text{Age}}\right)$$

where $\ln(\text{Yield})$ = natural logarithm of the biomass yield per unit area; b_i = regression coefficient, $i=0,1,2,\dots,6$; SI = site index (tree height in meters for white oak at base age 50 years); Age = stand age.

In most yield equations, stand age is expressed as a reciprocal to allow the asymptotic effect of yield with increasing age (Avery and Burkhart, 1994b). Due to heavy mortality in pine stands, the asymptotic number of trees (T_{asm}) for a certain basal area is determined according to the self-thinning rule, expressed as $\ln(T_{asm}) = a + b * \ln(Q)$, where Q is the quadratic mean DBH of all trees, and $\ln$ is the natural log function (Somers and Farrar Jr. 1991). For a given site index, models for the basal area and tree survival for natural longleaf pine stands showed exponential increase and decrease of stand basal area and T_{asm} , respectively, so that both stand characteristics approached their respective asymptotic values as stands aged (derived from Somers and Farrar Jr. 1991). The authors concluded that natural longleaf pine stands of high tree density that grow on good sites quickly approach T_{asm} where increase in stand basal area would only be possible if the heavy mortality is surpassed (Somers and Farrar Jr. 1991).

An important part of forest ecosystem dynamics is tree mortality (Harcombe 1987). In a review article based on data for 19 Lake States tree species, Harcombe (1987) reported that tree

mortality is more closely related to tree size than tree age, as shaded trees would have less energy to devote to survival and reproduction. For most ecosystems, concave mortality curves could be used to describe the decreasing risk of death with increasing age or size (Harcombe, 1987). However, some of the reviewed articles suggested a U-shaped death-rate curve, for the first part of which (declining mortality with increasing seedling size) is strongly supported by evidence, while the evidence for the latter part (mortality increasing at older ages (larger size classes) is weak (Harcombe, 1987). Harcombe (1987) concluded that shaded individuals in forests, due to the vertical structure, regardless of age, would have higher death rates compared to individuals that are exposed to light.

Leaf Litter and Root Biomass Decomposition Rates for Pine and Hardwood Stands

McClagherty et al. (1985) studied forest litter decomposition of six types of forest litter for two years in five adjacent forests located on Blackhawk Island in the Wisconsin River in south-central Wisconsin. The five forest types were sugar maple, bigtooth aspen, white oak, white pine, and hemlock. The hardwood stands were established on Alfisol soils, while the white pine stand was growing on a Spodosol soil, and the hemlock stand on a Histosol soil. All stands except the aspen stand were mature climax forests with no indication of recent disturbance (McClagherty et al. 1985). Their results indicated that the amounts of ash-free dry matter remaining after 1 year, estimated as percent of initial sample weight, were 51.07% and 56.2% for hardwood litter (average of estimates for sugar maple, aspen, and white oak litter) and for white pine litter, respectively (McClagherty et al. 1985). Their analysis showed that after 2 years of decomposition, the amounts of ash-free dry matter were 29.83% of the initial for hardwood litter, and 32.5% for white pine litter (McClagherty et al. 1985).

Silver and Miya (2001) used data from the literature to identify global and regional trends in root decomposition and root decay rates as a function of climate, substrate quality, environmental factors, and species groups, such as broadleaf, conifer, and graminoid species. Their summary indicated that buried litter-bags were the most common approach used for root-decay analyses. The authors reported that decomposition rates were similar for small (< 2 mm diameter) and medium (2-5 mm) roots, while decomposition of larger-diameter-roots (> 5 mm) was significantly slower (Silver and Miya 2001). Based on multiple regression analyses of root decomposition with environmental, climatic, and chemical parameters, Silver and Miya (2001) showed that actual evapotranspiration (AET), root Ca, and C:N ratio of root biomass explained 90% of the variability in root decay. Root Ca was reported to be the single best predictor of decomposition rates for roots; R-square of 33% when all roots were considered, and R-square of 55% for fine roots (Silver and Miya 2001).

Silver and Miya (2001) summarized that graminoid roots decayed the fastest, followed by broadleaf roots, and then coniferous species. Fitting the exponential decay function (Wider and Lang, 1982) to data from studies reporting mass loss (or carbon loss) over time, Silver and Miya (2001) developed root decomposition constants (k-values) for all-size roots by species groups: 0.44, 0.30, and 1.38 for broadleaf, conifer, and graminoid species, respectively. Using the exponential decay function ($y = y_0 * e^{-kt}$, where y_0 is the starting amount, and t is time) and these k-values, the time (in years) necessary for 95% of roots to decompose could be estimated as $t = -\ln(0.05)/k$, where $\ln$ is the natural log function: 6.81 years for broadleaf roots, 9.98 years for conifer roots, and 2.17 years for graminoid roots.

Burke and Raynal (1994) investigated fine root (< 3 mm) phenology, turnover, and decomposition rates in naturally regenerated hardwood forests located in the Adirondack State Park in New York, each dominated by one of the following species: sugar maple, red maple, American beech, and yellow birch. The soils were classified as Becket bouldery fine sandy loam (Typic Haplorthod) with high rock fragment content and organic matter, with distinct cool and moist conditions (Burke and Raynal 1994). The researchers measured root decomposition rates using standard mesh bag techniques. Both live and dead roots were collected with soil cores; roots were removed and cleaned from the soil, placed in 2-mm litter decay bags and buried 10 cm deep for 1 year. Their results indicated that the average percent root mass loss for the three root-size classes analyzed, < 0.5 mm, 0.5-1.5 mm, and 1.5-3 mm, was 38.07% after 1 year (Burke and Raynal 1994).

These studies showed that there is sufficient information in the literature that could be used to model litter and root carbon dynamics for hardwood and white pine stands. Although these results are established for undisturbed ecosystems, based on the finding by Insam and Domsch (1988), we believe that the reported litter layer and root decomposition rates could be used as reasonable approximations for modeling carbon accumulation in litter and root biomass on mined land. In a chronosequence study on reclaimed open pit-mined sites, managed as agricultural and forested land in the Rhine area in Germany and ranging from 1 to about 55 years old, Insam and Domsch (1988) reported that the microbial biomass increased to the levels of undisturbed soils in a relatively short period of time (about 15 years). However, in the same study, using the ratio of microbial biomass carbon to SOC as an indicator to evaluate the reclamation success, Insam and Domsch (1988) found that both chronosequences had not yet reached a steady state within 50 years of reclamation.

On some mined lands, because of the site-specific physicochemical characteristics of the soils, the amounts of decomposer microorganisms could be drastically reduced, thus leading to longer litter and root decomposition rates on mined sites compared to undisturbed ecosystems. In general, due to better soil conditions and energy source availability, microbial activity is greater on high-quality mined sites compared to poor-quality sites (Showalter et al. 2007).

Weed Control, Tillage, and Fertilization Effects on the Survival and Growth of Planted Trees on Mined Land

Weed Control Use on Mined Land. Because mine operators are required to establish complete vegetation cover soon after grading the mined land, they often use a mixture of aggressive and competitive grass and legume species (Torbert and Burger 2000). The resulting vegetation cover has proven to inhibit tree growth mainly because of the competition for light, water, and soil nutrients from the dense herbaceous vegetation stands (Ashby 1997; Skousen et al. 1994). Burger and Zipper (2002) proposed an entirely different revegetation procedure that they called “forestry reclamation approach” to distinguish it from traditional grassland establishment. They developed and recommended a tree-compatible erosion control seed mixture that competed less with the seedlings.

In a study of 15 abandoned mined land sites, Skousen (1994) concluded that if the post-mining land use chosen by the land owner is forestry, then different reclamation approaches should be used. Instead of planting the site to dense herbaceous vegetation stands, Skousen (1994) suggested that nurse logs, tree debris, or soil ridges may be used for soil erosion control and to provide suitable conditions for tree growth and forest establishment. In the case of one of

their study sites, herbaceous vegetation cover seemed to have delayed forest regeneration for 30 years (Skousen et al. 1994).

Another prominent mined land researcher tested the effects of weed control by applying herbicides before tree planting on a mined land in southern Illinois (Ashby 1997). The results after years of growth showed a significant average increase in height growth for all 16 tree species in the study. The overall height response was 21%, where the average height was measured 117 cm for non-treated plots and 142 cm for herbicide-treated plots (Ashby 1997). Additionally, ripping and the interaction between herbicide treatment and ripping on tree height growth were tested. The overall data analysis indicated that the combined effect of ripping and herbicide treatment significantly increased tree height growth (+84% versus control), compared to ripping-only (+42% versus control), herbicide-only (+20% versus control), and the control (no ripping and no-herbicide treatments) (Ashby 1997).

Tillage Use on Mined Land. With the passage of SMCRA in 1977, among other provisions, mine operators were required to produce stabilized and visually pleasing, smooth surfaces on reclaimed mined lands. However, the excessive compaction of soils by trafficking heavy mining machinery inhibited the survival and growth of planted trees (Ashby 1997; Torbert and Burger 1990; Torbert and Burger 1996; Zeleznik and Skousen 1996).

Torbert and Burger (1990) reported significant decrease in tree survival on graded and tracked-in soils compared to rough-graded sites on a partially reclaimed mined land site in Wise County, VA. Percent survival ranged from 0 (sycamore) to 63% (Virginia pine) on graded sites, and ranged from 42% (Virginia pine) to 100% (sugar maple and black walnut) on rough-graded sites (Torbert and Burger 1990). Virginia pine was the only tree species that survived better on compacted sites than on rough-graded sites. However, the results showed that for all measured tree species, black walnut, northern red oak, sugar maple, sycamore, Virginia pine, and white pine, planted seedlings grew taller on rough-graded sites than compacted sites. The height response ranged between 0% (red oak) and 100% (sugar maple) (Torbert and Burger 1990).

In another study near Lovely, KY, Torbert and Burger (1996) reported similar results for the survival of tree species on moderately-graded and intensively-graded sites. The survival rate and the height increase were both greater on moderately-graded sites compared to intensively-graded sites. For the same study, Torbert and Burger (1996) ripped (also referred to as sub-soiling) a previously graded mined site to simulate the surface soil conditions that may be created by rough-grading, eliminating the compaction effect of mine soils. Their results indicated that ripping significantly increased the survival of three (sweetgum, yellow-poplar, sycamore) of the five tree species tested, when compared to intensively-graded sites, and significantly increased the survival of two of the species (sweetgum and yellow-poplar), when compared to moderately-graded sites (Torbert and Burger 1996). White pine and loblolly pine tree species were reported to be unaffected by compaction intensity, although the survival of loblolly pine increased by approximately 15%, from 15% to 32% to 43%, by reducing the compaction level from intensively- to moderately-graded to ripped sites, respectively (Torbert and Burger 1996).

Fertilization Use on Mined Land. Nilsson (1993) reported that above-ground carbon accumulation rates of a 30-year-old Norway spruce stand, growing on poorly developed Spodosols, increased by 65% after 3 years, compared to a control, following irrigation combined with liquid fertilization with 100 kg ha⁻¹ yr⁻¹ N. The effect of fertilization in these Norway

spruce stands was reported to still be positive at year 5 of the treatment. For another study, Johnson et al. (2003) reported that fertilization effect was no longer significantly affecting tree growth at age 18 in a loblolly plantation study.

Canary et al. (2000) reported an average of $34.7 \text{ Mg C ha}^{-1}$ increase of forest ecosystem carbon stock following N fertilization at approximately $70 \text{ kg ha}^{-1} \text{ yr}^{-1}$ N over 16 years in a Douglas-fir plantation in western Washington. The authors reported that the forest floor carbon stock was significantly increased due to fertilization (+34%,) but no SOC increase was significant over the 16-year period. However, upon deeper soil sampling, Canary et al. (2000) found out that greater amount of SOC were accumulating in deeper horizons, where 40% of the 0-85 cm carbon was found below 25 cm depth.

In an extensive review of the literature regarding fertilization on mined land, Rodrigue (2001, his Chapter II) concluded that fertilization does not increase tree survival, but could increase tree growth depending on the mined land conditions. Some of the reviewed studies showed that fertilization increased tree growth only during the first growing season, while others reported about 12% tree height growth 10 years after tree planting (Rodrigue 2001). Rodrigue (2001, his Chapter II) summarized that some fresh mine soils could release nutrients (due to rock weathering) sufficient for tree growth, however, some mine soils could be N- and P- limiting. Generally, between 50 and 75 kg N ha^{-1} in the early stages of stand development to ensure good tree survival, and greater-than- 100 kg P ha^{-1} to ensure a long-term supply of soil P are recommended for mined lands (Rodrigue 2001).

Despite the positive results of forest fertilization on the increase of carbon sequestration in terrestrial ecosystems, Schlesinger (1999) cautioned that fertilization for carbon storage may not lead to actual net carbon sequestration in terrestrial ecosystems. The costs of fertilizer production are commonly ignored in carbon sequestration studies. For every mole of N-fertilizer produced, 0.375 moles of carbon are emitted in the atmosphere from CH_4 used for fertilizer production (Schlesinger 1999). For example, for the production of 1 kg of diammonium phosphate fertilizer (molecular weight equal to 132 g), $3.4091 \times 10^{-2} \text{ kg}$ of elemental C, estimated as $0.375 \times 12/132$, will be emitted in the atmosphere from burning fossil fuels. Therefore, nearly 1 Mg of elemental carbon from fossil fuels would be emitted in the atmosphere by applying the fertilizer C-cost values for a one-time fertilizer application on 100 ha of reforested mined land.

Ramped Cultural Treatments for Three Forest Types. Due to the somewhat site- and species-specific responses to weed control, tillage, and fertilization, as described above, a research group at Virginia Tech established a three-state field trial illustrated in Figure 1-5-2. This study aimed to research the effect of different intensities of forest establishment treatments, weed control (WC), weed control plus tillage (WC + T), and weed control plus tillage plus fertilization (WC+T+F), on the survival and growth of three species groups, white pine, hybrid poplar (hybrid 52-225, *P. trichicarpa* x *P. deltoides*), and a mix of hardwood species (Fig. 1-5-2). Casselman et al. (2006) reported the first growing-season tree survival and growth of planted trees on reclaimed mined lands for the three study sites located in Ohio (OH), West Virginia (WV), and Virginia (VA). Prior to stand establishment, along with other chemical and physical soil properties, soil samples from these study sites were collected and analyzed for SOC content. The spoil types ranged from sandstone (in VA), to shale (in WV), and siltstone (in OH) (Fig. 1-5-2) (Casselman et al., 2006). In this study, Casselman et al. (2006) applied weed control and pre-emergent herbicides prior to planting in order to eliminate the competition from previously

sown grasses and legumes. Herbicide was also used to eliminate grass competition after the first year during which seedlings were shielded from the applied herbicide.

Tillage plots, assigned according to the experimental design, were ripped to depth of 61 to 91 cm (Casselman et al. 2006). Although tillage usually leads to lower soil bulk density, which in turn improves root growth, the authors recommended that tillage be a site-specific management practice dependent on the properties of the mine spoil (Casselman et al., 2006). On one hand, due to the platy structure of shale rock fragments, root growth was greatly limited by the compacted shale spoil (Casselman et al. 2006). For the mined sites in WV, where Casselman et al. (2006) reported shale spoil of relatively high bulk density (1.68 g cm^{-3}) with no topsoil layer, tillage increased significantly the tree growth and survival. On average, among the tree species investigated in WV, tree survival increased from 51 to 69%, and first-year tree height growth increased from 8.8 to 24.1 cm due to tillage (Casselman et al. 2006). Although tillage was expected to improve tree survival and growth on siltstone-derived mine soils in OH, similar to the fine-textured shale-derived mine soils in WV, the specific soil properties in OH caused contrasting results among the tree species groups tested. The mined site in OH had a topsoil layer underlain by a siltstone-derived spoil of relatively high density (bulk density = 1.7 g cm^{-3}) leading to standing water conditions, and impeded root growth (Casselman et al. 2006). While tillage increased the survival of hardwood species from 60 to 72% and slightly reduced it for hybrid poplar (from 49 to 45%), its effects resulted in a relatively substantial survival decrease among white pine seedlings, from 45 to 29% (Casselman et al. 2006). The average tree growth was not significantly affected by tillage, although the reported tree growth on tilled plots was 17.4 cm, and was 13.3 cm on non-tilled plots.

On the other hand, tillage had no significant effect on tree growth and survival on the VA mined sites. These sites were developed from oxidized sandstone spoil of relatively high compaction (bulk density = 1.74 g cm^{-3}); however, due to the coarser texture of these soils, comprised predominantly of sand-sized particles, as opposed to silt- and clay-sized particles, root growth was not impeded by the spoil's density (Casselman et al. 2006). On average, tree survival was 77% and 71%, and tree growth was 16.9 and 25.1 cm, on tilled and non-tilled plots, respectively (Casselman et al. 2006).

In general, Casselman et al. (2006) reported that tree growth and survival on sandstone spoils was greater than that observed on siltstone and shale materials. Among other properties, sandstone materials have lower pH (more acidic) and lower electrical conductivity (EC), which are good for tree growth, thus making oxidized sandstone a superior tree growth material (Casselman et al. 2006). This study indicated that soil tillage on mined land, particularly mined land with fine-textured soils, could greatly increase the survival rate of planted tree species and, thus, can increase the number of potentially well-growing trees which will sequester carbon in vegetation biomass, litter layer, and the soil.

In the same study, fertilization plots received 272 kg ha^{-1} of diammonium phosphate delivering 49 kg N ha^{-1} , $55.1 \text{ kg P ha}^{-1}$, added via banded application, and $46.8 \text{ kg K ha}^{-1}$ along with a suite of other micronutrients (S, B, Cu, Mn, and Zn), all of which were applied at the base of each seedling after seedlings were planted. In general, Casselman et al. (2006) recommended weed control be applied, as it was shown to always have a positive effect on tree growth and survival.

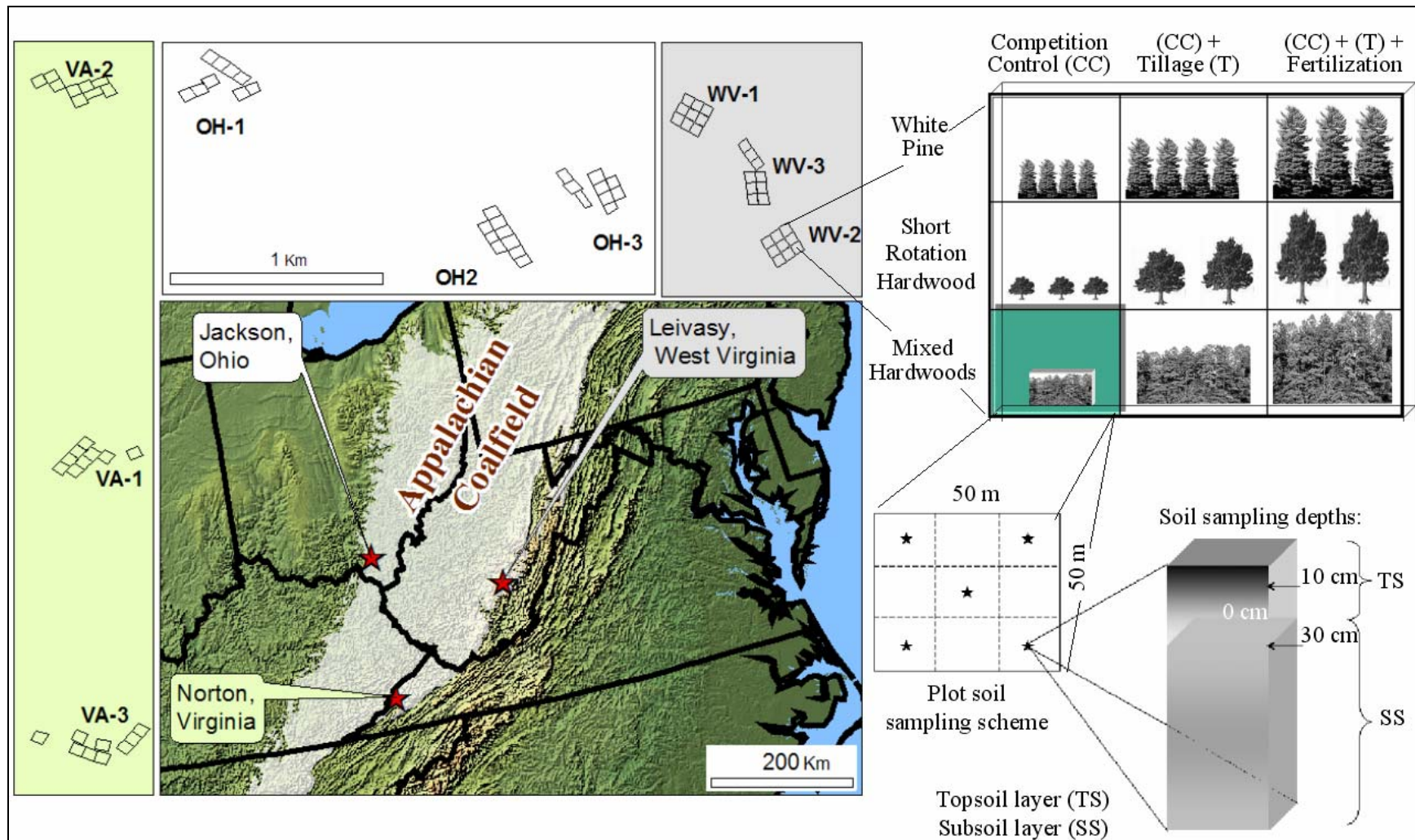


Figure 1-5-2. Field location, spatial distribution of treatment blocks, and soil sampling scheme for 9 study sites located in Virginia, West Virginia, and Ohio. The 9-cell schematic (top right corner) represents a 3 x 3 factorial experimental design (3 cultural treatments x 3 forest types) of the research project that was installed in the winter of 2003.

On their OH mined sites, because of dense vegetation cover of competitive grasses (e.g., fescue) and legumes, fertilization resulted in significantly lower tree survival due to stimulated growth of the competing grasses (Casselman et al. 2006). On average, only 14% of all planted tree seedlings survived on the WC+T+F plots, compared to 49% on their WC+T plots (Casselman et al. 2006). It was also suggested that adding fertilizer may increase the soluble salts in these siltstone-derived mine soils, thus increasing the soil water potential of the soil solution, which could lead to increased moisture stress and seedling mortality (Casselman et al., 2006). The mine spoil on this site had neutral pH (6.9) and relatively high salt content, evident from the high values of soil electrical conductivity ($EC = 0.47 \text{ dS m}^{-1}$), both of which were very close to the ranges of pH and EC known to impede tree growth (Casselman et al. 2006). On average, tree growth was not affected significantly by the fertilization treatment.

Similar to the sites in OH, fertilization of the shale-based WV mine soils decreased tree survival to 43%, compared to 69% observed on their WC+T plots (Casselman et al. 2006). The authors reported similar tree growth of live seedlings following fertilization on the WV sites, 23.7 cm and 24.1 cm for their WC+T+F, and WC+T plots, respectively (Casselman et al. 2006). However, fertilization increased tree growth on the sandstone-derived mined lands in VA, from 25.1 (WC+T plots) to 46.7 cm (WC+T+F plots), because this sandstone material was low in N and had a high P-fixing capacity (Casselman et al. 2006).

The first growing-season findings for the effect of tillage on tree survival and tree growth increase (decrease) reported by Casselman et al. (2006) agreed with other mined land studies (Ashby 1997; Rodrigue 2001; Torbert and Burger 2000)(Fig. 1-5-3). Rodrigue (2001, his Chapter II) did an extensive review of the literature to summarize the results from mined land studies that reported the effect of tillage on tree survival and growth. The age of the mined lands used in the reviewed studies ranged from 2 to 46 years; however, there was no data for the 10-30 and 30-46 year ranges (Figure 1-5-3). On average, tree survival was higher on ripped mine sites (about 61%) compared to unripped (graded) sites (about 48%). As expected, tree growth was also increased due to soil ripping; on average, percent tree height increase due to soil ripping was reported at approximately 59%.

Clearly shown in Figure 1-5-3, the tree survival and growth data reported by Casselman et al. (2006) were within the range of data for older mined sites that ranged in spoil rock type and soil conditions. Therefore, we believe that it is logical to incorporate the findings by Casselman et al. (2006) on mine soil tillage into the first iteration of the DSS, as proposed in this paper.

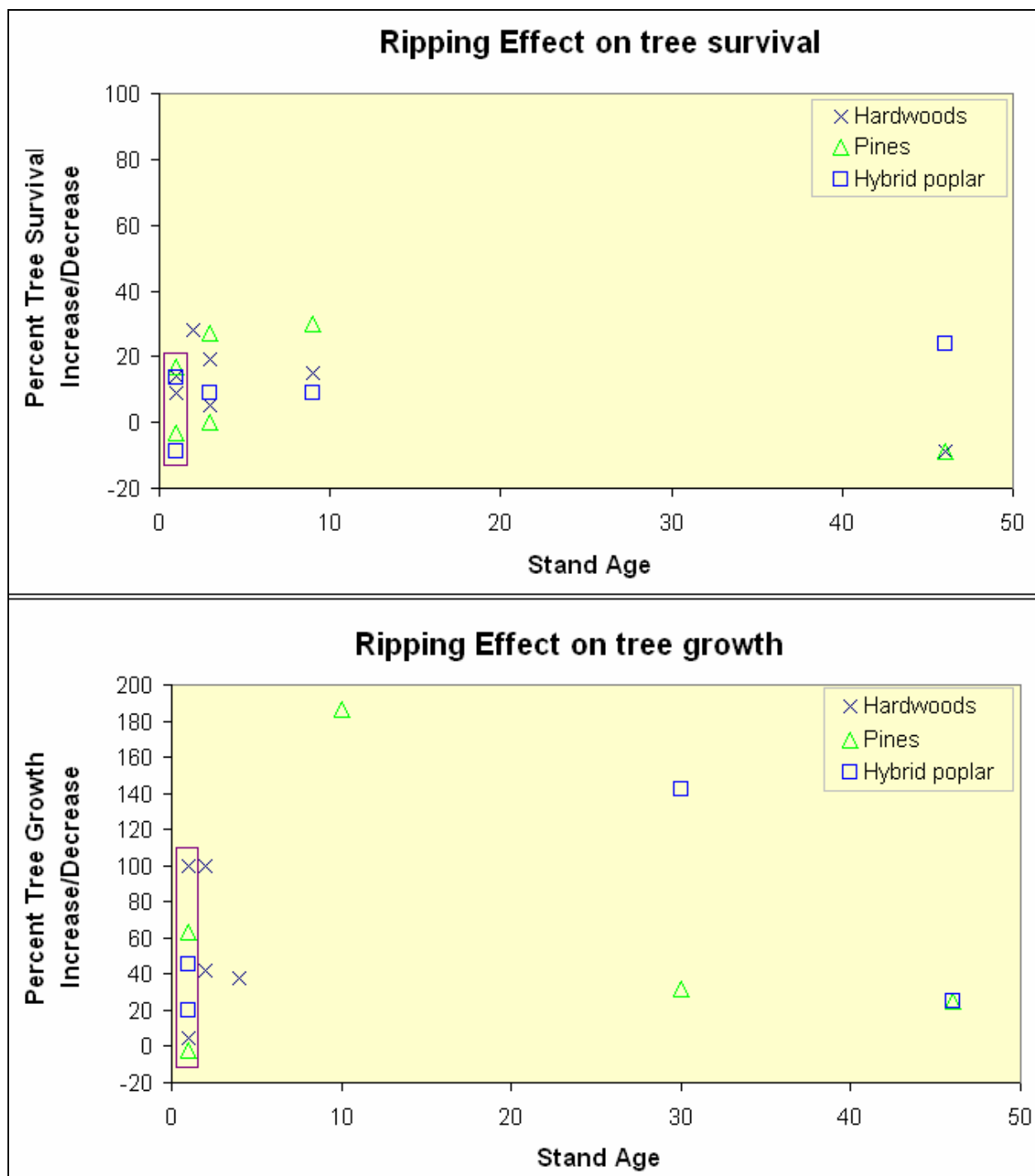


Figure 1-5-3. Hardwood, pine and hybrid poplar tree survival and growth increase/decrease due to soil ripping on mined land. The rectangle depicts the first growing season results reported by Casselman et al. (2006).

Forest Management Effects on Changes of the SOC Pool

In a summary article, Johnson (1992) reported that most forest management practices, such as forest harvesting, site preparation, and fertilization had no significant effects on soil carbon content flux. The reported SOC changes (+/- 10%), following harvesting, were only somewhat varying, where some studies reported large net losses and others reported net gains. Johnson (1992) also reported varying results following prescribed burning where low-intensity fires were usually associated with small changes in SOC, and high-intensity fires and wildfires could result in large losses.

Because soil organic carbon inputs are directly related to changes in above-ground biomass production – that is, as above-ground biomass increases, the amount of litter layer, deposited on the soil surface, would increase, and the cumulative below-ground tree biomass production, as coarse- and fine-root biomass, would increase – it could be expected that above-ground biomass increase will always lead to SOC increases. However, Johnson et al. (2003) showed that after 18 years following the establishment of a loblolly pine plantation near Oak Ridge, TN, no statistically significant changes in SOC were found, while the ecosystem carbon stock tripled in size, from 54 to 161 Mg C ha⁻¹. They reported that 93% of the net ecosystem carbon was allocated in the biomass and 6% was in the forest floor.

There is paucity of information in the literature regarding the effects of forest management on carbon stock changes on mined lands. Existing reports about the carbon sequestration rate of mine soils vary widely. Akala and Lal (2001b) reported that mine soils in Ohio sequestered atmospheric CO₂ at rates ranging from 0.5 to 3.1 Mg C ha⁻¹ yr⁻¹ for the surface 0-15 cm soil depth for sites with pasture post-mining land use, and at rates from 0.7 to 2.3 Mg C ha⁻¹ yr⁻¹ (0-15 cm depth) for sites with forestry post-mining land use. However, the authors point out the possibility of erroneous measurements in SOC content due to coal and shale contamination in the mine soil evident from the observed high variation in the measured rates of carbon sequestration in these Ohio mine soils (Akala and Lal, 2001b). Other researchers reported much lower carbon sequestration rates for mined land grasslands ranging from 0.1 Mg ha⁻¹ yr⁻¹ (in western North Dakota) to 0.3 Mg ha⁻¹ yr⁻¹ (in western Montana and Saskatchewan) (Wali 1999).

Varying by mined site quality and intensity of forest management practices, soil carbon will eventually reach an equilibrium level between 110 and 150 years following forest establishment (Akala and Lal 2001b). At equilibrium, there will be a balance between the carbon sequestered in the soil and the carbon dioxide emitted back to the atmosphere due to soil microbial activity, hence, there will be no further net carbon sequestration in the mine soil.

Carbon Storage in Wood Products in Use and in Landfills, and Associated Greenhouse Gas Emissions from biomass Burning and Fossil Fuel Usage during tree Biomass Harvesting, transportation, Wood Product Manufacturing, and Recycling

Because of the technological advances in the forest industry, namely in the areas of tree biomass harvesting, wood product manufacturing, and the proper disposal of wood products in carefully designed landfills, it is now possible to estimate the net amount of sequestered carbon in wood products after the trees are harvested (Liski et al. 2001). Although carbon storage in wood products in use and in landfills does not imply the same process of carbon sequestration occurring during photosynthesis in tree leaves, some of the carbon locked in wood biomass, and protected by lignin, could remain in landfills indefinitely (Micales and Skog 1997; Miner 2003;

Skog and Nicholson 1998), thus alleviating the effect of atmospheric CO₂ on global climate change. In contrast, there are many intermediate steps of harvested roundwood transformation into final products and product-usage, at which energy from burning fossil fuels is used for wood product manufacturing, starting from tree biomass harvesting and ending with permanent carbon sequestration in landfills (Karjalainen and Asikainen 1996; Liski et al. 2001). Therefore, a thorough and realistic modeling approach, including modeling fossil fuel usage for harvesting, transportation, manufacturing, and recycling (Liski et al. 2001) is critical to determine the amount of net sequestered carbon in wood products.

In their summary report for the year 2000, the Intergovernmental Panel on Climate Change (IPCC) reported 3,100 million m³ world production of roundwood, representing about 700 Tg C (1Tg = 10¹² g = 10⁶ Mg) (IPCC, 2003). Of this amount, approximately 20% were reported to remain as long-term manufactured wood products: 420 million m³ as sawtimber, representing about 90 Tg C, 220 million m³ as wood-based panels and fiberboard, representing about 50 Tg C; and about 480 m³ as wood for pulp for paper, which was used to produce about 230 million tonnes of paper and paperboard, representing about 150 Tg C (IPCC, 2003). The results from this report present the magnitude and importance of carbon storage in wood products, which should be studied, analyzed, and reported in detail, as is the amount of sequestered carbon in forest ecosystems.

Carbon Emissions from Wood Harvesting, Transportation, Manufacturing, and Recycling. Along with achieving greater work efficiency, mechanization of the work at each step of the forestry and wood production industries has dramatically increased the fossil fuel usage in forestry. Karjalainen and Asikainen (1996) reported the energy use from fossil fuels in the forest industry in Finland. In particular, they reported the emissions of greenhouse gases (GHGs) from fossil fuel use in silvicultural and forest-improvement work, wood harvesting operations, and wood transportation. The total amount of GHGs, including CO₂, N₂O, CO, CH₄, NO_x, and non-methane volatile organic compounds (NMVOC), emitted from fossil fuel use during land scarification, i.e., tillage and disking, was equivalent to 33.49 x10⁻³ Mg ha⁻¹ of fuel C, after adjusting for the global warming potential (GWP) of each greenhouse gas (Karjalainen and Asikainen, 1996). Similarly, in pine stands, approximately 12.04 x10⁻³ and 6.80 x10⁻³ Mg fuel C per 1 Mg C in harvested biomass were emitted in the atmosphere from harvesting and biomass haulage during thinning and final harvesting operations, respectively. These estimated were relatively lower in hardwood stands, in which approximately 9.85 x10⁻³ and 5.56 x10⁻³ Mg fuel C per 1 Mg C in harvested biomass were emitted from harvesting and biomass haulage during thinning and final harvest (Karjalainen and Asikainen, 1996).

The amount of GHG emissions during long-distance transportation of harvested timber was about two orders of magnitude lower than the harvesting and haulage emissions. Karjalainen and Asikainen (1996) reported that approximately 6.29 x10⁻⁵ and 5.14 x10⁻⁵ Mg fuel C per 1 Mg C in transported biomass were emitted during loading, transportation, and unloading of pine and hardwood biomass, respectively. Karjalainen and Asikainen (1996) estimated 94 km to be the average distance to a timber or a pulp and paper mill in Finland, where the forest industry is a significant part of the economy.

Liski et al. (2001) studied the effects of the choice of rotation length on the net carbon stock in tree biomass, soils, and wood products, as well as the amount of fossil fuels used for harvesting and manufacturing of wood products. Different scenarios for managing Norway spruce and Scots pine forests in Norway were analyzed. For their model, the authors used actual

data of raw wood material division into production lines, such as sawmill, plywood mill, and pulp-and-paper-mill, including chemical, mechanical and recycled paper production (Liski et al., 2001). At final harvest of a forest managed for sawtimber, 44% of the cut roundwood actually became sawtimber, 44% was processed for pulp and paper products, and 12% was nonprocessable residue which was burned for energy (Liski et al. 2001). Furthermore, 50% of the sawtimber was used in residential buildings, 15% as changeable construction material, 10% for furniture, 15% as construction aid material, and 10% as packing material. The authors also reported that during product manufacturing, for each 1 Mg C in processed sawtimber biomass, 0.032 Mg of fossil fuel C were released in the atmosphere from burning fossil fuels for energy, such as electricity (Liski et al. 2001).

Similarly, at final harvest of a forest managed for pulpwood, on average, 69% of the roundwood was processed for pulp and paper products, and 31% was burned for energy as a nonprocessable residue. Thirty-three percent of the pulpwood material was used for books and office supplies, 33% as packing material, and 34% for newspapers and magazines (Liski et al. 2001). The authors reported carbon emissions of approximately 0.305 Mg of fossil fuel C for each 1 Mg C in processed pulpwood, which is nearly one order of magnitude greater than the fossil fuel costs associated with sawtimber processing (Liski et al. 2001). The highest fossil fuel costs were reported during product recycling; carbon emissions approximated 0.48 Mg C for each 1 Mg C in recycled material (Liski et al. 2001). Furthermore, the authors reported that only 83% of the material collected for recycling was manufactured back into pulp and paper, and 17% was burned for energy as nonprocessable residue.

Liski et al. (2001) concluded that in Scots pine forest management, long rotation lengths were most favorable, in which largest carbon stocks were accumulated with the smallest fossil fuel emissions. Similarly for Norway spruce stands, Liski et al. (2001) recommended that long rotations, with smallest fossil fuel emissions and slightly smaller forest carbon content (compared to that at shorter rotations) be chosen for carbon sequestration. The reported downside of choosing long rotations was the delay of revenues for the land owners (Liski et al. 2001).

Similar results were reported in an evaluation study of loblolly plantation management regimes for wood products and carbon sequestration for the Chesapeake Forest located on the lower eastern shore of Maryland (Spinney et al. 2005). The regimes with longest rotation lengths, 35 and 40 years with two intermediate thinnings, were reported to yield the highest discounted revenues from roundwood production, as well as the highest carbon stocks (Spinney et al. 2005).

Carbon in Wood Products In Use. The half-life of carbon in a certain type of wood product is the time over which one-half of the carbon in the original material leaves the carbon pool of wood products in use (Miner 2003). The Intergovernmental Panel on Climate Change has considered the use of a simple first-order relationship to convert the half-life of carbon into a decay curve that allows scientists to calculate the fraction of carbon remaining in wood products in-use as a function of time (Miner, 2003). The first-order decay curve is represented by the following equation (Miner, 2003):

$$FR = \left(\frac{1}{1 + 0.69315 / HL} \right)^Y$$

where FR = fraction of carbon remaining in use in year Y, HL = half-life in years, and Y = elapsed time in years.

The amount of carbon remaining sequestered for long periods of time in forest products can be significant. In a summary report, Miner (2003) showed that the annual sequestration of carbon in forest products ranged from 26 to 139 Tg C yr⁻¹, which could be a sufficient amount to offset the GHG emissions from forest product manufacturing. Skog and Nicholson (1998) estimated that currently, additions to wood and paper products in use and in landfills in the United States, including net imports, sequester about 61 Tg C yr⁻¹, after deducting for CO₂ and CH₄ emissions from burning and biomass decay in landfills.

Skog and Nicholson (1998) reported the half-life of carbon for widely-used wood products in the United States. The average half-life of carbon in residential buildings, for both single- and multi-family homes, was estimated to be 85 years, and 16 years for carbon in furniture, 6 years for carbon in palettes and for carbon in books and office supplies, and 1 year for all other paper products, including packing material, newspapers and magazines (Skog and Nicholson 1998).

Carbon in Wood Products In Landfills. The ultimate destination of most wood products, following their time in use, is the landfill. Wood and paper products sent to dumps prior to 1986 included solid wood residues, construction and demolition waste, paper and paperboard products (Skog and Nicholson, 1998). A portion of these materials was burned and their contents were exposed to oxygen leading to more complete decay; i.e., the carbon sequestered in these wood products was returned to the atmosphere as CO₂, or as a more potent greenhouse gas – CH₄, which is known to have approximately 21 times the global warming potential of CO₂. However, new legislation required that dumps be phased out by 1986, and wastes be placed in landfills (Skog and Nicholson, 1998). Landfills are periodically sealed, which prevents oxygen from entering the system.

In a detailed study investigating the decay processes occurring in landfills, Micales and Skog (1997) showed that "the placement of forest products in landfills serves as an important carbon sink, and its importance in the global carbon balance should not be overlooked." Oxygen that is still available under the landfill capping soil layer is rapidly used by white-rot fungi, which can decay lignin to a certain extent, thus slowly creating an O₂-limiting environment that, in effect, promotes fermentative metabolism for certain bacteria (Micales and Skog, 1997; Skog and Nicholson, 1998). After oxygen is depleted, only anaerobic bacteria remain, which are not capable of decaying lignin, but only the exposed cellulose and hemicellulose molecules. The fraction of cellulose and hemicellulose that is encased in lignin remains out of reach of the bacteria and thus can remain undecomposed permanently (Micales and Skog, 1997; Skog and Nicholson, 1998). For example, 20% to 27% of newsprint is lignin, making newspaper very resistant to decay (Skog and Nicholson, 1998). Their calculations showed that only 0-3% of the carbon from wood and an average of 26% of carbon from paper is potentially released into the atmosphere as CO₂ and CH₄, once the material has been landfilled (Micales and Skog, 1997).

More detailed results about the amount of permanently sequestered carbon in landfills is presented by Miner (2003). As one of the primary subjects of their report, the authors discussed the significance of carbon sequestration in landfills and the offsetting effects of CH₄ emissions from landfills. Landfill wood product decomposition data are very limited due to difficulties associated with conducting any studies investigating landfill dynamics and the factors that can influence wood decomposition rates in municipal solid waste landfills. Miner (2003)

summarized that moisture content, pH, and the inherent degradability of the landfill material were the most important factors in landfills regarding wood product decomposition.

The resulting body of research indicated that lignin was essentially non-degradable under anaerobic conditions, and due to the physicochemical characteristics of wood products such as the presence of inorganic coatings and lignin, the cellulose and hemicellulose were isolated, making much of the carbon in these materials inaccessible to the microorganisms that would otherwise decompose them (Miner, 2003). For modeling studies, because landfill decay constants are not available for the degradable portion of landfilled C, Miner (2003) proposed a method assuming that none of the degradable carbon is sequestered in landfills, even though it is known that a certain fraction of it could remain in landfills for very long periods of time. Miner (2003) assumed that only the non-degradable carbon would be sequestered in the landfill environment. The non-degradable carbon was expressed as a fraction of the total carbon using carbon storage factors (CSFs) for different materials, the data for which were developed from laboratory experiments lead by the U.S. Environmental Protection Agency (EPA) (Miner, 2003). The CSFs for widely-used wood product types in the United States are as follows (Miner, 2003): 0.58 for packing material (corrugated cardboard), 0.85 for magazines (coated paper), 0.93 for newspapers (uncoated paper made up of mechanical pulp), 0.11 for office paper (uncoated, made up of primarily bleached pulp fibers), and 0.85 for branches, wood, and solid-wood products.

Skog and Nicholson (1998) reported that 40% of the degradable portion of wood products in landfills was emitted as CO₂, and 60% was emitted as CH₄. The estimated half-life of CO₂ emitted from landfills was approximately 3 years, and was 20 years for CH₄ (Skog and Nicholson, 1998). This is to say that one-half of the carbon that would be released from landfills as CO₂ and CH₄ gases would move out of the landfill within 3 and 20 years, respectively. Micales and Skog (1997) reported that, after methanogen bacteria are established in landfills, methane production can last between 8 and 40 years. Eventually methane production stops and the landfill becomes an inert mass (Micales and Skog, 1997). It has been shown that 10% of CH₄ is oxidized to CO₂ by soil microorganisms inhabiting the cover soil as it moves out of the landfill (Micales and Skog, 1997).

Methods and Materials

Dynamic Systems Modeling in STELLA® Environment of Forest Carbon Sequestration on Mined Land

We used total tree carbon prediction models for white pine and native hardwood stands developed in Substudy 4 and hybrid poplar growth-models from the literature to estimate the carbon content in forest stands of the respective forest types. These regression models were incorporated into the DSS (Fig. 1-5-4) developed within a dynamic systems modeling environment, in which some of the input variables could be managed by the DSS user, as described in later sections of this substudy.

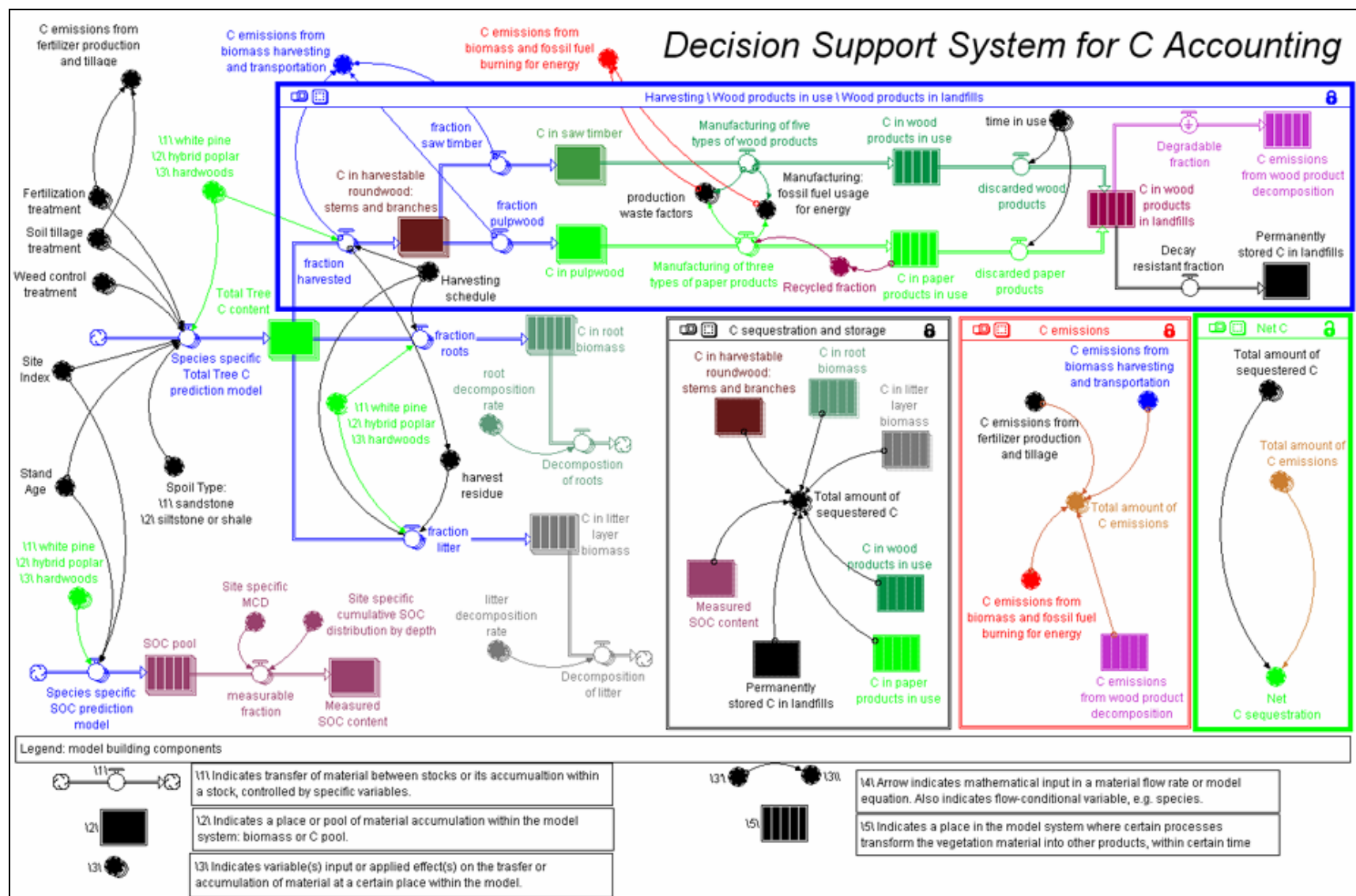


Figure 1-5-4. Structure of the proposed decision support system tool based on expert knowledge for predicting carbon sequestration for forest establishment practices on mined land. Net amount of carbon sequestration is estimated by subtracting the *Total amount of C emissions* from the *Total amount of sequestered C*, shown in the rightmost rectangle within the model structure.

We used the STELLA® dynamic systems software (Research Version 7.0.2 for Windows, High Performance Systems, Inc., Hanover, NH) to accommodate the interactive variable input for forest establishment practices, including choice of mine spoil type, choice of fertilization and/or tillage treatments, thinning and final harvest scheduling, the type and amount (as fractions) of forest products generated from harvested roundwood at the end of the rotation, to mention just a few. Figure 1-5-4 shows the components and the base structure of the proposed DSS for modeling the amount of carbon sequestered in mined land forest ecosystems, the carbon stored in wood products produced from harvested roundwood from these forest ecosystems, and the carbon emissions from biomass and fossil fuel burning for energy.

We allowed the DSS-user to interactively select the appropriate intensity of forest establishment practices, such as weed control, weed control and soil tillage, and weed control and soil tillage and fertilization. Based on these site-specific selections and their cumulative effect on the survival and tree growth of three forest types (white pine, hybrid poplar, native hardwoods), we estimated the net amount of sequestered carbon in each component of the forest ecosystem, total tree biomass, litter layer, root biomass, and the soil, as well as the carbon amount stored in wood products in-use, and in-landfills (Fig. 1-5-4).

By incorporating known information about tree biomass allocation to stems, branches, roots, and leaves for the three modeled forest types, as well as using litter and root decomposition rates reported in the literature, we modeled the actual carbon stocks accumulated in these ecosystem components as a function of time. Based on SOC models for hybrid poplar stands reported in the literature, and based on our findings in Substudy 4 regarding SOC accumulation under pine and hardwood vegetation, we modeled soil organic carbon accumulation in the soils of the respective forest ecosystems (Figure 1-5-4). Although some mine soils could sequester significant amounts of carbon, only a fraction of that amount could be measured cost-effectively for carbon accreditation (Substudy 2); integrating these findings, we estimated the cost-effectively measurable fraction of the total soil organic carbon stock.

An important phase of the carbon cycle in a forest is the harvesting of tree biomass for wood product manufacturing. We enabled the DSS-user to enter the number of harvesting events (thinnings and final harvest), and their respective scheduled times (indicating stand age), during which roundwood would be cut and manufactured into wood and paper products. We also provided the option for the DSS-user to input their best-known estimates for the amount of harvested forest biomass that would be allocated as saw timber and pulpwood at the end of the rotation (30 years for white pine, 60 years for hardwoods) (Fig. 1-5-4). Because the longevity of pulpwood products is often dramatically different compared to sawtimber products, we recommend that allocation of harvested biomass to timber and pulpwood be carefully selected. And yet, selecting sawtimber and pulpwood fractions must also be valid for the specific tree species groups that are analyzed. For example, fast growing hybrid poplar may be more suitable for pulpwood production than timber, and harvested white pine is more valuable as a timber producing species than pulpwood. By making the harvest-scheduling and roundwood allocation to sawtimber and pulpwood interactively controlled by the DSS-user, we allowed mined land owners from different parts of the country, with differing local timber and pulpwood markets, to choose the most likely harvesting regime, thus indicating the most likely fate of sequestered carbon locked in the wood (Fig. 1-5-4).

Finally, we predicted the carbon emissions from fossil fuel usage for energy (e.g., electricity and fuel for machinery operation), biomass burning for energy, and biomass

decomposition, each depicting carbon emissions along the wood production framework of the forest industry (Figure 1-5-4). We modeled the carbon emissions from fossil fuel usage for fertilizer production and tillage, tree biomass harvesting, roundwood transportation, wood product manufacturing, biomass burning and fossil fuel burning for energy, and wood product decomposition in landfills. To estimate the net carbon sequestration following forest establishment practices on mined land, we subtracted carbon emissions due to forest industry operations from the cumulative carbon stock sequestered within each forest ecosystem component, including tree stems, branches, litter layer, root biomass, and measured SOC content, the carbon in wood and paper products in use, and the permanently sequestered carbon in landfills (Figure 1-5-4).

Below, we present detailed descriptions of each modeling component used in our proposed DSS: (i) development of hybrid poplar carbon sequestration models (total tree C, litter layer C, and SOC), (ii) white pine carbon sequestration models (total tree C, litter layer C, and SOC), (iii) hardwood carbon sequestration models (total tree C, litter layer C, and SOC), (iv) modeling the effects of forest establishment practices on tree growth and survival on mined land, and (v) modeling carbon storage in wood products and carbon emissions from forest establishment practices, tree harvesting, biomass transportation, wood product manufacturing, and biomass burning.

Hybrid Poplar Carbon Sequestration Models

We adopted the diameter-growth-by-age model proposed by Wu et al. (2003), which we assumed to depict hybrid poplar growth on high quality sites, SI of 45 m. The predicted DBH was then used to estimate tree stem carbon content using the tree biomass equations reported by Netzer et al. (2002) and the assumption that 50% of tree biomass is elemental carbon. In order to convert the tree stem carbon from Mg units to units representing a certain forest area (Mg ha^{-1}), we assumed that hybrid poplar trees would be planted at 2.4 x 3 m spacing, for a total of 1345 trees ha^{-1} (adopted from Casselman et al., 2006), and tree mortality in these stands would be similar to that of *P. tremuloides* (quaking aspen) stands (derived from Harcombe, 1987). Mortality rate as fraction of live trees within a DBH-size class ranging from 4 to 37 cm was approximated by developing a regression equation based on mortality (dependent variable) and DBH (dependent variable) data in Harcombe (1987); the resulting regression equation was as follows: $\text{Mortality Rate} = \text{Exp}[(-0.75337 + 0.00401 * (\text{DBH,cm})^2 - 0.23917 * (\text{DBH,cm}))]$, ($n=34$, $P < 0.0001$, $R^2 = 98.83$, $\text{Adj.-}R^2 = 98.76$). For trees of DBH-size class outside the 4-37 cm range, we assumed negligent mortality. The resulting S-shaped, hybrid poplar growth function by stand age depicting total tree carbon content, and the number of live trees per hectare in a forest stand on high quality sites are presented in Figure 1-5-5.

Based on the findings by Hibbs and Bower (1998), we estimated tree stem carbon content for any site quality class with SI ranging from 15 to 45 m, using the stem carbon predictions for mined lands of SI 45 m. For example, assuming that biomass yield of hybrid poplar decreased 5.8% for every 1-m decrease in SI (Hibbs and Bower, 1998), we calculated that a 5-year-old hybrid poplar stand growing on land of SI 44 m would accumulate $22.04 \text{ Mg C ha}^{-1}$ in stem biomass, which would be 5.8% less than a similar stand growing on mined land of site index 45 m with $23.40 \text{ Mg C ha}^{-1}$ contained in the stem biomass.

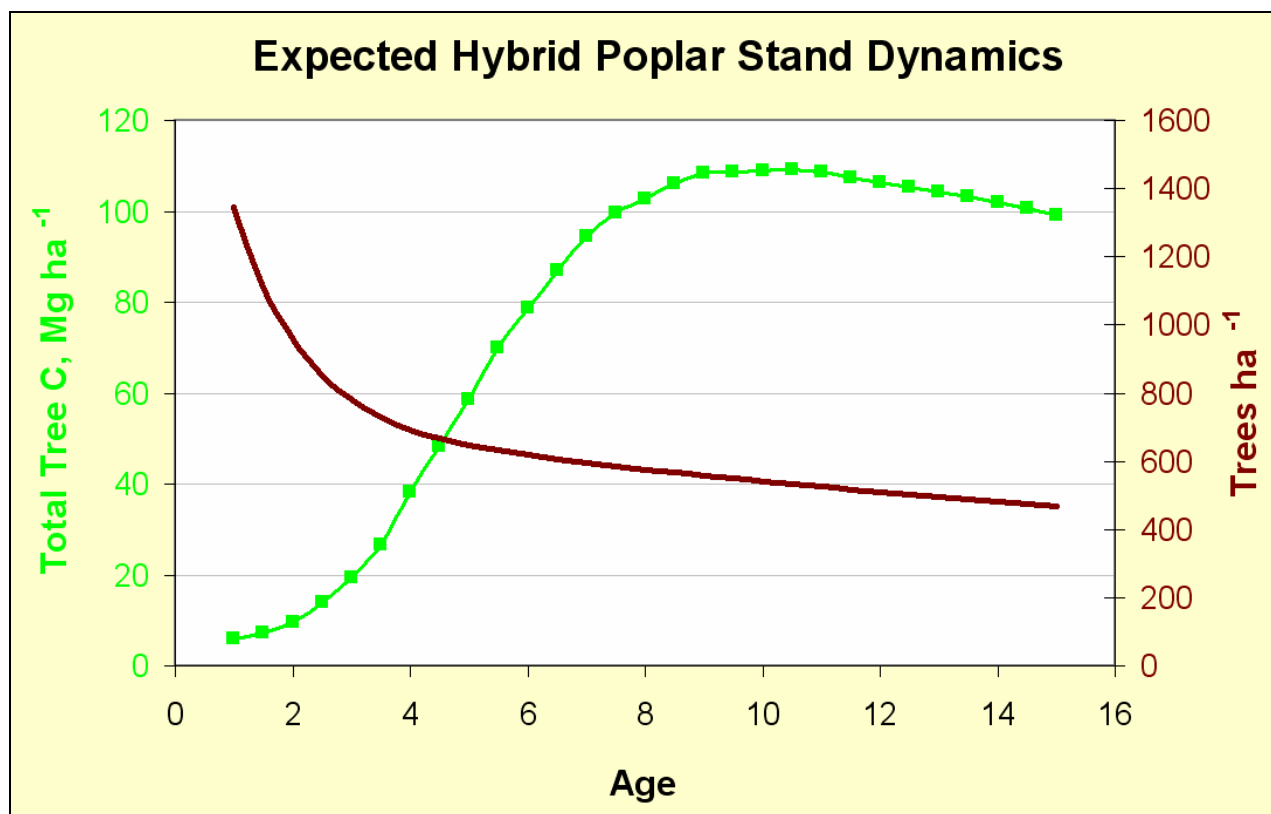


Figure 1-5-5. Expected hybrid poplar growth function and tree survival of forest stands planted at initial 2.4 x 3 m tree spacing (1345 trees ha⁻¹) on tilled and fertilized undisturbed lands of high site quality (SI=45 m); tree mortality was modeled as described in the text.

To create a continuous surface across stand age and site index depicting hybrid poplar total tree carbon accumulation, we used the biomass distribution equations reported by Wullschleger et al. (2005) to estimate total tree carbon from stem carbon content. For example, based on the reported relationship between stem biomass and total tree biomass of measured poplar trees, $(Total\ tree\ biomass, g/tree) = [(stem\ biomass, g/tree) / 0.2787]^{(1/1.0298)}$ (Wullschleger et al., 2005), a 5-year-old hybrid poplar stand growing on forest land of site index equal to 44 m would sequester 55.23 Mg C ha⁻¹ in total tree biomass, estimated as $[(22.04\ Mg\ C\ ha^{-1}) * (2\ Mg\ biomass\ Mg^{-1}\ C) * (10^6\ g\ Mg^{-1}\ biomass) / (648\ trees\ ha^{-1})] / 0.2787^{(1/1.0298)} * 648 / 2 / 10^6$ (Figure 1-5-5). The continuous surface depicting the expected total tree carbon accumulation across stand age and SI is depicted in Figure 1-5-6b. For modeling purposes, we developed a best-fit response surface, based on a mathematical function with stand age and SI as independent variables that would predict total tree carbon content under hybrid poplar forest vegetation. The resulting regression model was based on 899 values from the expected hybrid poplar growth surface and was of relatively high R-square (96.55%) (Figure 1-5-6a).

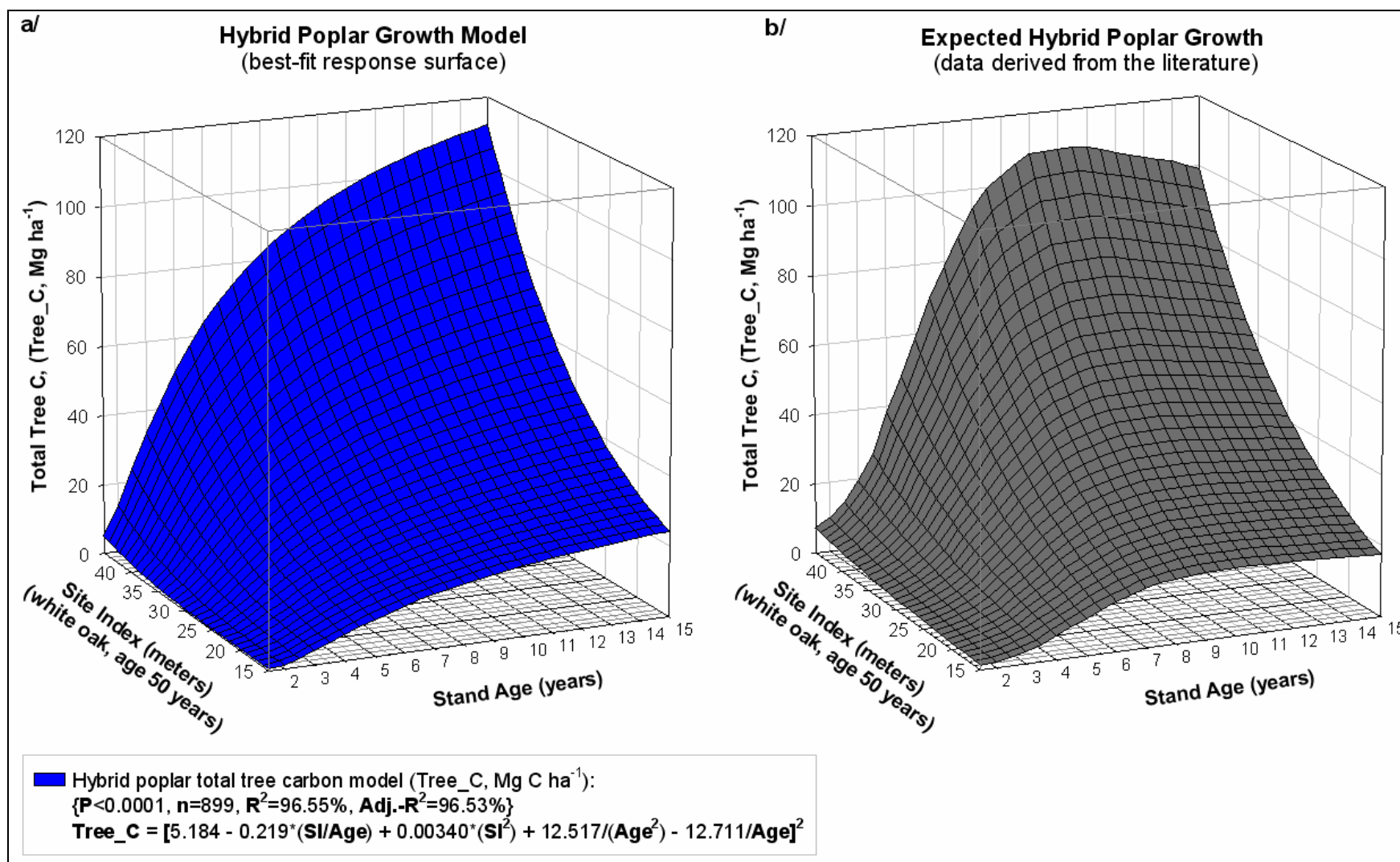


Figure 1-5-6. Best-fit total tree carbon response surface (a/) as a function of SI and Age for hybrid poplar stands developed by fitting a regression equation to the tree growth data (b/) derived from the literature for tilled and fertilized stands established on non-mined land. We assumed initial tree density of 1345 trees ha⁻¹ and modeled tree mortality as described in the text.

We used the total tree biomass distribution equations reported by Wullschleger et al. (2005) to model the carbon content in each of the forest biomass components. The cumulative carbon content in tree stems and branches was used as an input to the *Harvesting wood products in-use and wood products in-landfills* section of our DSS (Figure 1-5-4). At harvesting, we assumed that 15% of the harvestable biomass, including stems and branches, would remain on the site as harvest residue, which would become part of the litter layer carbon pool. Because of the nature of tree harvesting, we realize that the amount of harvest residue left on the site is a factor of forest health, site topography, harvesting intensity, to mention just a few; therefore, we provided the option for the DSS user to adjust the level of harvest residue to a more realistic value, based on the user's expertise.

The leaves and root biomass distribution equations (Wullschleger et al., 2005) were used to model the litter layer and root carbon pools in our DSS. Based on the findings by Cotrufo et al. (2005), we assumed that leaves and branches (as harvest residue), entering the litter layer in a hybrid poplar stand would be completely mineralized within a period of approximately 2 years. Similarly, we assumed a 2-year-period for the time necessary for dead roots (fine and coarse) to be completely mineralized in the mine soil (adopted from Coleman et al., 2000).

We used the SOC regression equation by Hansen (1993) to model the net SOC accumulation in mine soils under hybrid poplar vegetation as a function of stand age. As indicated in the equation, in order to estimate net SOC content, a correction for litter layer carbon was necessary (Hansen, 1993). We calculated the amount of litter layer carbon as a function of total tree biomass (Wullschleger et al., 2005), assuming that 50% of biomass is elemental carbon.

To determine the measurable fraction of the total SOC content, we modeled the cumulative SOC distribution of a forested mine soil developed under a white pine forest stand growing on a weathered sandstone spoil in Wise County, Virginia (Fig. 1-5-7). We measured soil organic carbon down the soil profile as percent by sample weight using the 16-step method, described in Substudy 1. The SOC analysis was done in the fine soil fraction (< 2 mm) of soil samples collected at the centers of distinct soil layers down the profiles of 14 deep soil pits. We assumed that the physical properties of the site, such as bulk density and coarse rock fragment content, remained unchanged for the 26-year period of stand development, and were similar to those measured on a nearby 2-year old mined site that was planted to grass.

We assumed that the MCD of the forested mine soil is 50 cm, the default input in our DSS, meaning that about 85% of the total profile (0-200 cm) SOC content would be measurable (Fig. 1-5-7b). However, we allowed the option for the DSS user to enter a more realistic MCD estimate, based on the soil properties of the mined site of interest. Also, we allowed the option for the user to modify the graphical input describing the cumulative SOC distribution on the site, from its current setting for a 26-year-old white pine forest, presented in Figure 1-5-7b, to a precisely analyzed SOC distribution for the mined site of interest.

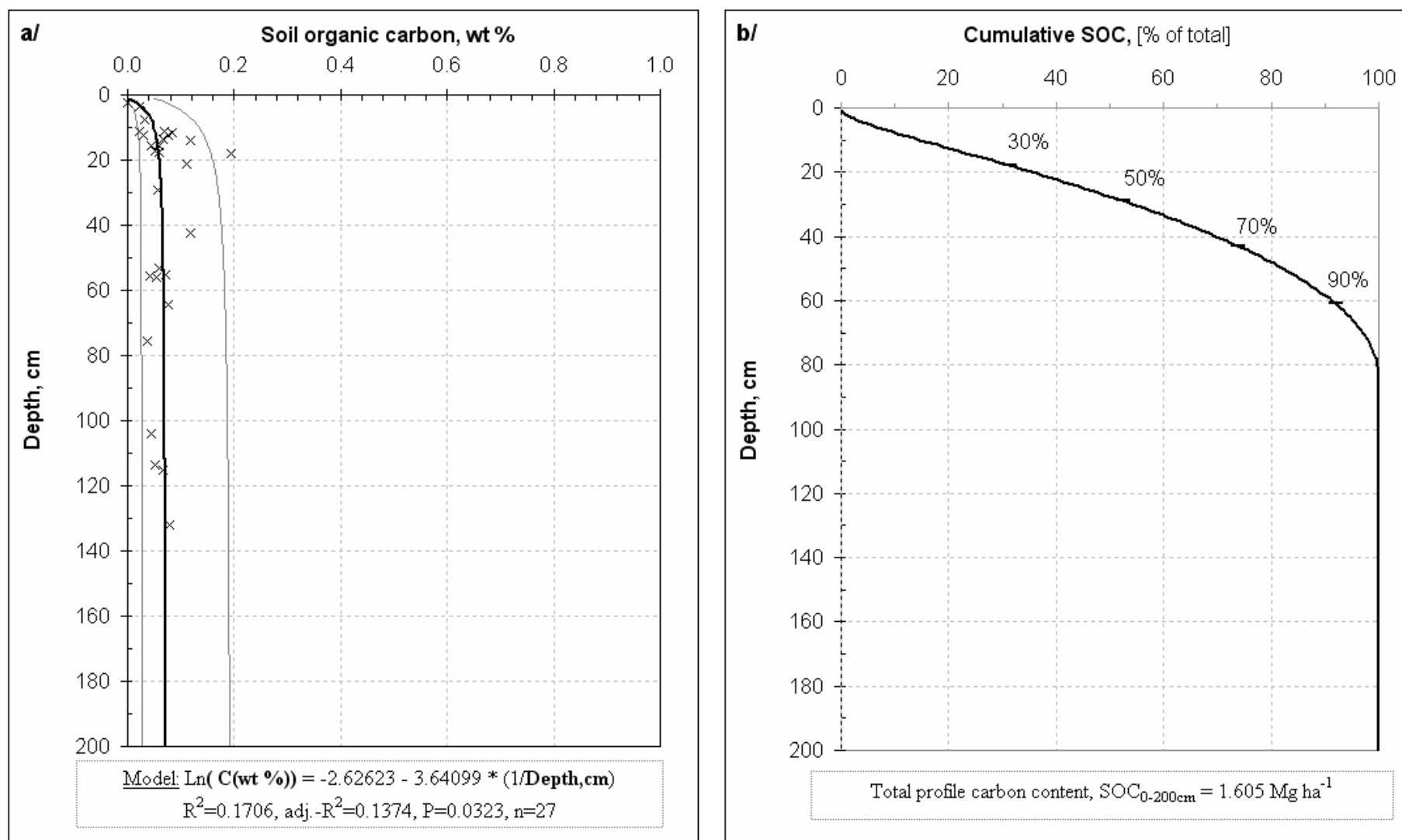


Figure 1-5-7. Soil organic carbon concentration model (weight percent) by depth (a/) and cumulative SOC distribution by depth (b/) in a mine soil derived from weathered sandstone and supporting a 26-year old white pine forest stand located in Wise County, Virginia. The dark line in a/ represents the model mean SOC(wt%) by depth and the light lines represent the 95% confidence limits of the mean.

Both the MCD and the cumulative SOC distribution are required inputs in our DSS in order to predict the measurable fraction of the total sequestered SOC. Based on the findings by Akala and Lal (2001a) and Heath et al. (2002), we assumed that the SOC in mine soils under forest vegetation would continue to accumulate until the equilibrium level of undisturbed forest is reached, after which there would be no further increase in SOC content. The regression equation used to estimate the equilibrium SOC level of the undisturbed forests adjacent to the mined sites studied in Substudy 4 was as follows: ($P < 0.01$, $n = 31$, $R^2 = 22.31\%$, $\text{Adj.-}R^2 = 19.63\%$): $\text{SOC}_{0-200 \text{ cm}}, \text{Mg ha}^{-1} = \text{Exp} [1.659 + 0.509 * (\text{SI}^{1/2})]$.

White Pine Carbon Sequestration Models

We developed total tree carbon sequestration models, as a function of mine soil quality, measured by site index, and stand age, for white pine and hardwood trees growing on forested mined lands using data reported in the literature. Because the plot-level data used for our analysis in Substudy 4 (Figs. 1-4-5 and 1-4-6) were based on sampling plots that contained both pine and hardwood species, we used individual tree measurements in each plot to model tree growth across the spectrum of SI and age. We queried the vegetation biomass data from Rodrigue (2001) to create two separate data sets for all measured white pine and hardwood trees at each surveyed forested mined plot. Vegetation biomass data for the respective tree species were extracted from 55 mined land measurement plots to model total tree carbon content across SI and age. Using total tree and tree component biomass equations from the literature (Jenkins et al., 2003), we computed the total tree carbon content of measured trees of the two species groups, thus producing two carbon data sets, which we used for separate regression analyses regressing total tree carbon content by site index and stand age in SAS® statistical analysis software (SAS, 2007).

Except the hardwood stands identified in Substudy 4, none of the surveyed plots were located in pure forest stands, but rather in forest stands of mixed pine and hardwood species composition. We developed plot-area correction factors for each white pine and hardwood tree within each surveyed plot to properly determine the forest area occupied by the individual trees, which would be a fraction of the measurement plot area. We assumed that the extent of the measurement plot area (0.04048 ha) occupied by an individual tree was proportional to the basal area (BA) of that tree, estimated as $\text{DBH}^2 * \pi / 4$ in cm^2 units, where π (pi) = 3.141593. For example, for white pine (WP) trees growing together with other tree species, including other pines and hardwood trees, the plot-area correction factor was computed as $BA_{WP} / BA_{all \text{ trees}}$, where BA_{WP} is the cumulative basal area of the white pines within a plot, and $BA_{all \text{ trees}}$ is the total tree basal area on that plot. This factor was used to determine the total tree carbon content of all white pine trees representing the proper forest area within the plot (Mg C ha^{-1}) as follows:

$$\text{Tree\_}C_{WP}, \text{Mg ha}^{-1} = \left(\frac{\text{Tree\_}C_{WP}, \text{Mg}}{0.04048, \text{ha}} \right) * \left(\frac{BA_{all \text{ trees}}, \text{cm}^2}{BA_{WP}, \text{cm}^2} \right)$$

where $\text{Tree\_}C_{WP}$ = tree carbon content sequestered by all white pine trees within a plot; 0.04048, ha = area of the measurement plots used by Rodrigue (2001) in ha units; $BA_{all \text{ trees}}$ = basal area of all trees measured in the plot in cm^2 units; BA_{WP} = basal area of all white pine trees measured in the plot in cm^2 units.

We performed linear regression analyses to develop total tree carbon models as a function of site index and stand age for white pine and hardwood forest stands. The regression model was of relatively high R-square, 79% (Figure 1-5-8). The extent of the data, which ranged between 18 and 40 years, encompassed the period when timber would most be likely harvested (i.e., rotation age 30 years), indicating that reasonable rotation age predictions from this model were possible. Rotation age 30 is usually used in pine stand management.

Model predictions outside the extent of the data were also reasonable. Total tree carbon content increased exponentially with the increase of SI (Fig. 1-5-8). Across the spectrum of stand age, the model predictions described a well-known and biologically sound pine stand dynamic in unmanaged pine stands; in particular, the observed high stand mortality of pine trees as they age beyond 25 (30) years. This dynamic was more prominent for sites of higher site quality. The white pine dynamic described by the model in Figure 1-5-8 supported the findings by Vimmerstedt (1962). Vimmerstedt (1962) used data (age, SI, tree spacing, biomass yield) from 112 sample plots in unburned and unthinned white pine plantations in the Appalachians, which he used to develop a regression model ($R^2 = 90\%$) to predict biomass yield in unmanaged white pine stands. Due to the above reasons, it was logical to use the total tree carbon model in Figure 1-5-8 to predict the carbon sequestration by white pine stands established on mined land.

We used the tree component biomass equations reported by Jenkins et al. (2003) to estimate the carbon content in tree stems, branches, leaves, and roots. Because the aboveground and component biomass equations were reported for individual trees as a function of DBH, we calculated the expected average tree DBH based on the total tree carbon predictions. The total tree carbon content data by Rodrigue (2001) depicted in Figure 1-5-8, the reported relationship between aboveground biomass and DBH for pines, $aboveground\ biomass\ (kg\ tree^{-1}) = Exp(-2.5356 + 2.4349 * Ln(DBH, cm))$ (Jenkins et al., 2003), and a tree density prediction (Figure 1-5-9), were all used to estimate the expected average DBH in white pine stands as follows:

$$DBH(cm) = Exp \left[\frac{Ln \left(\frac{TotTreeC, Mg\ ha^{-1} * 1000 * 2 / (Trees, ha^{-1})}{1 + 0.22} \right) + 2.5356}{2.4349} \right]$$

where $TotTreeC, Mg\ ha^{-1}$ = predicted total tree carbon content in $Mg\ ha^{-1}$ units; multiplication by $(1000*2)$ is a unit conversion factor to convert carbon content in $Mg\ ha^{-1}$ units to biomass content in $kg\ ha^{-1}$ units, assuming that 50% of biomass is carbon; $Trees\ ha^{-1}$ = tree density per hectare (Figure V-10); division by $(1+0.22)$ is based on the assumption that the root biomass for conifer trees with DBH up to 50 cm is approximately equal to $0.22*aboveground\ biomass$ (derived by Jenkins et al., 2003), hence, $total\ tree\ biomass = aboveground\ biomass*(1+0.22)$. Based on the above assumption, carbon content in aboveground biomass would be equal to $(TotTreeC, Mg\ ha^{-1}) * 100/122$, and carbon content in root biomass would be equal to $(TotTreeC, Mg\ ha^{-1}) * 22/122$.

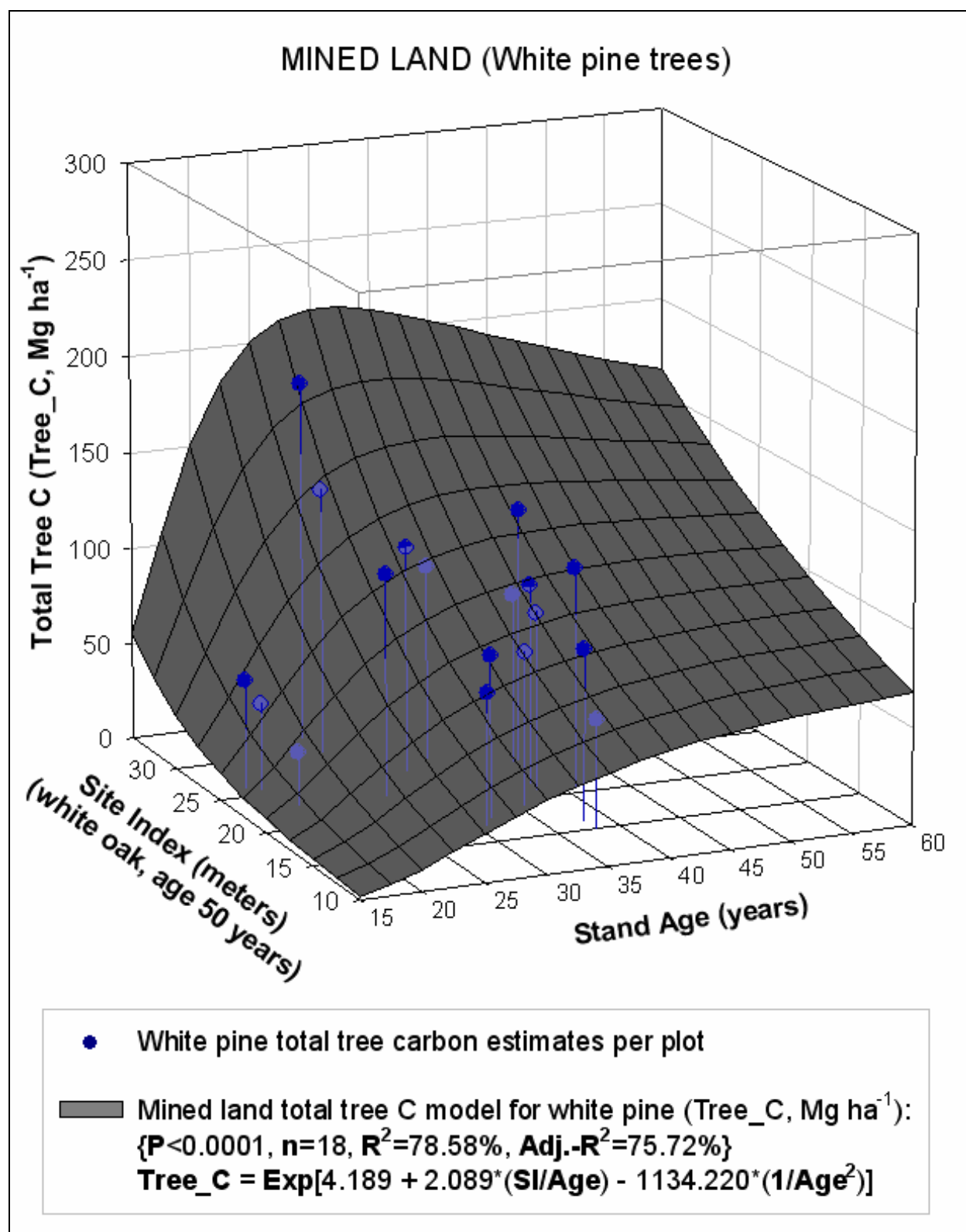


Figure 1-5-8. Total tree carbon content prediction model for white pine forest stands established on mined lands in the Midwestern and Appalachian coalfields. These data were extracted from a larger data set reported by Rodrigue (2001) and were analyzed using the methods described in the text.

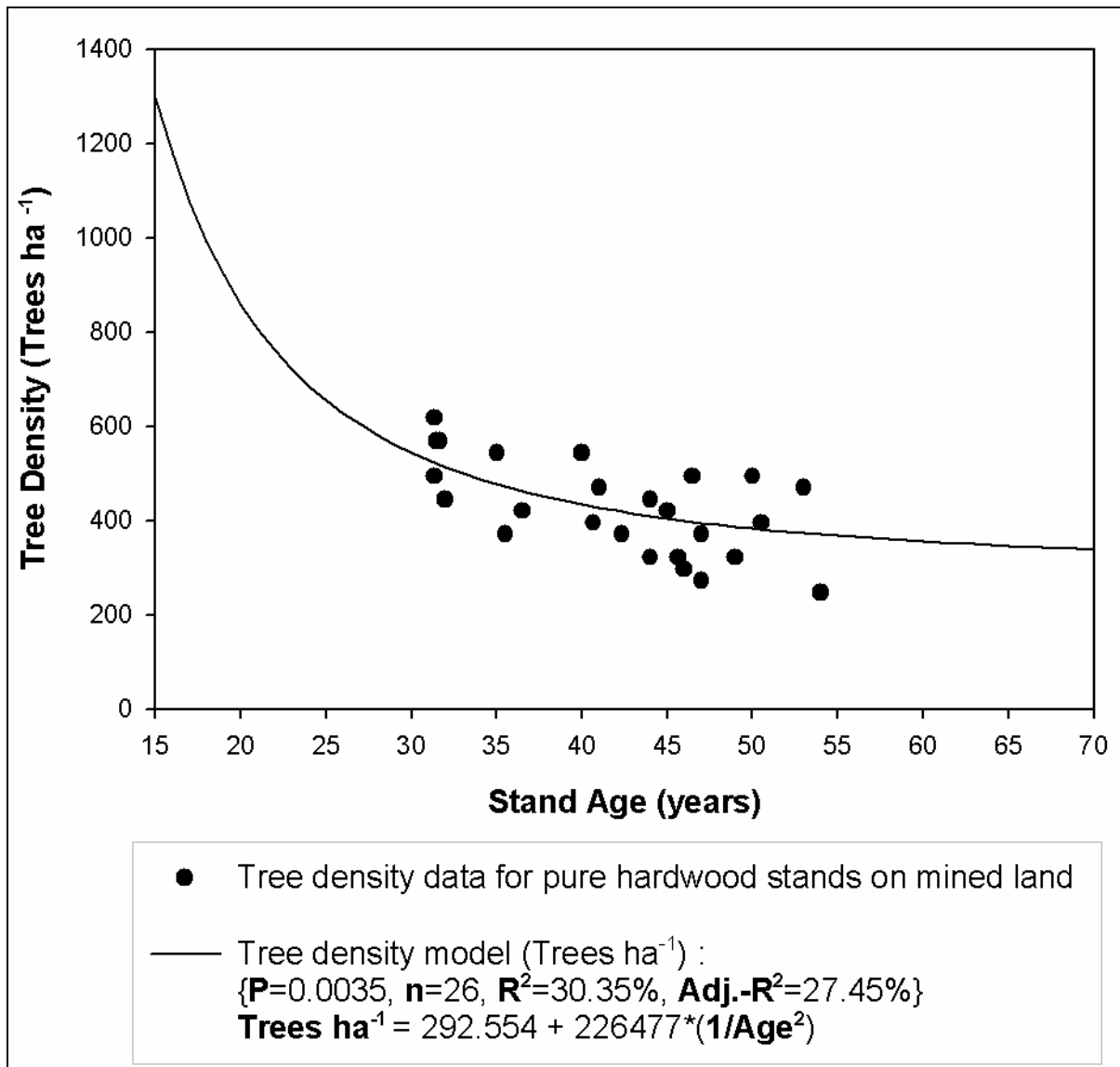


Figure 1-5-9. Measured tree density data for pure hardwood stands established on mined lands (Rodrigue, 2001), and a regression model describing the approximate tree densities in these stands.

We used the number of measured trees in each plot of the hardwood stand data set by Rodrigue (2001) to develop a regression equation that would predict tree density as a function of stand age; the regression equation was as follows: $\text{Trees ha}^{-1} = 292.554 + 226477 \cdot (1/\text{Age}^2)$ (Figure 1-5-9). This equation was only used to estimate the expected average DBH as depicted above. However, because there was no tree density information in the data set by Rodrigue (2001) that could be used to determine tree mortality in forests on mined land, we assumed that white pine tree mortality as fraction of live trees within a DBH-size class in undisturbed forests (Harcombe, 1987) could be used to approximate tree mortality in white pine stands on mined land. We developed a regression equation for DBH classes ranging from 9 to 48 cm based on mortality (dependent variable) and DBH (dependent variable) data in Harcombe (1987); the resulting regression equation was as follows: $\text{Mortality Rate} = \text{Exp}[(-5.44880 + 0.03041 \cdot (\text{DBH,cm})^{1/2} + 24.77601 \cdot (1/\text{DBH,cm}))]$, ($n = 22$, $P < 0.0001$, $R^2 = 95.79$, $\text{Adj.-R}^2 = 95.34$).

For trees of DBH-size class outside the 9-48 cm range, we assumed negligent mortality. If stand thinning treatments were selected (described in later sections), then we assumed that tree mortality would be negligible prior and after the thinning, in which case, the number of live trees per hectare was computed as the difference between initial tree stocking and the number of trees removed during each thinning.

Using the expected average DBH estimate, we computed the total tree carbon content (Mg ha^{-1}) for future stands that could be established at land owner-specified tree spacing that could differ from the tree density graph in Figure V-10. Total tree carbon content was estimated as $\text{TotTreeC (Mg ha}^{-1}) = (1+0.22) * ((\text{Exp}(-2.5356 + 2.4349 * \text{Ln}(\text{DBH,cm})) / 1000) * (\text{Tree\_density}))$, where *Tree density* is the expected number of live trees per hectare when trees are planted at 2.4 x 3.0 m tree spacing for a total 1345 trees ha^{-1} (adopted from Casselman et al., 2006) and predicted mortality rate based on the findings by Harcombe (1987).

We estimated the fraction of total tree carbon stored within the remaining tree components, stem, leaves, and branches as follows (Jenkins et al., 2003): stem (bark + wood), $\text{Mg C ha}^{-1} = (\text{TotTreeC, Mg ha}^{-1}) * (100/122) * (\text{Exp}(-2.0980 - 1.1432/\text{DBH}) + \text{Exp}(-0.3737 - 1.8055/\text{DBH}))$; leaves, $\text{Mg ha}^{-1} = (\text{TotTreeC, Mg ha}^{-1}) * (100/122) * (\text{Exp}(-2.9584 - 4.4766/\text{DBH}))$; and branches (Mg ha^{-1}) were computed as the difference $(\text{TotTreeC, Mg ha}^{-1}) * (100/122) - (\text{stem, Mg C ha}^{-1}) - (\text{leaves, Mg C ha}^{-1})$. The leaves and root biomass distribution equations were used to model the litter layer and root carbon pools in our DSS. We assumed that the litter layer biomass in white pine stands would be decomposing at rates similar to the findings by McClaugherty (1985); that is, approximately 32.5% of litter layer biomass would remain after 2 years.

In order to model carbon content accumulation in root biomass, we assumed that white pine roots (coarse and fine) would decompose and completely mineralize in approximately 10 years, based on the findings by Silver and Miya (2001). To implement this in our DSS, we used available modeling objects (i.e., *conveyor* object with transit time set to 10 years) in STELLA® software that allowed time delays to be incorporated within the model.

Because thinning of forest stands is a powerful tool that foresters often use to manage overstocked stands and salvage tree biomass that could be lost as tree mortality, as well as to allocate land resources to desired tree species for high quality timber (Nyland, 1996), we implemented thinning options in our DSS. We allowed the user to select an appropriate stand management regime from three available options: no thinnings, one thinning (scheduled between stand age 17 and 23 years), and two thinnings (the first being carried out between stand age 17 and 23 years, and the second - between 23 and 28 years). For each of the thinning scenarios, the rotation age could be set between stand age 30 and 40 years.

If thinning treatments were selected, we allowed the user to select the appropriate thinning intensity, represented by percent trees to be removed during each thinning (we assumed a default of 30%), and the expected tree growth increase of the remaining trees due to greater resource availability and space opening during each thinning operation. We assumed 70% increase in tree growth with allowed a range between 0 and 250 %, based on suggested diameter increment increase of 1 to 6.5 times the previous increment following thinning to various residual stocking levels (Chapter 19 in Nyland, 1996).

We assumed that the biomass removed during thinnings would be used for pulp and paper production. We assumed that approximately 15% of harvestable biomass (stems and branches) removed during thinnings would remain on the site and become part of the litter layer carbon

pool. The harvesting residue would then begin to decompose and mineralize due to microbial activity in the forest floor and the soil. We assumed that the harvesting residue from thinning of white pine stands would be decomposing at rates similar to the litter layer biomass.

Soil organic carbon accumulation under white pine vegetation was modeled from empirical data reported by Rodrigue (2001) for forested mined sites in the Midwestern and Appalachian coalfields (Substudy 4). The analyses of these data, representing pine forests established on reclaimed loose mine spoils, indicated that mean SOC content accumulation to 2-m depth after 60 years could be predicted by site index using the following equation ($P = 0.05$, $n = 12$, $R^2 = 32.66\%$, $\text{Adj.-}R^2 = 25.93\%$): $\text{SOC}_{0-200 \text{ cm}}, \text{Mg ha}^{-1} = [1.653 + 0.00443 * (SI^2)]^2$. This result was based on the assumption of a steady rate of SOC accumulation through the years of stand development. We assumed a linear relationship between SOC content and stand age in order to predict SOC with time. Based on the findings by Akala and Lal (2001a) and Heath et al. (2002), we assumed that after a certain period of time, SOC content would reach a new equilibrium level, approximately that of undisturbed forests, after which there would be no further increase in soil carbon content. Similar to the modeling approach for hybrid poplar stands, we used the equilibrium SOC level of the undisturbed forests, adjacent to the mined sites studied in Substudy 4. We also assumed that the MCD of the forested mine soil is 50 cm, meaning that about 85% of the total profile (0-200 cm) SOC content would be measurable (Figure 1-5-7b).

Hardwood Carbon Sequestration Models

Similar to our analysis for individual white pine trees on mined land described above, we developed a total tree carbon sequestration model for hardwood trees as a function of site index and stand age, based on the vegetation biomass data from Rodrigue (2001). The resulting linear regression model was used to predict total tree carbon content in forest stands ranging between 15- and 75-years old (Fig. 1-5-10). As expected, the predictions at rotation age 60 made by this model were nearly identical to the rotation-age, total tree carbon content predictions depicted in Figure 1-4-7. Although the fit of the regression model based on individual tree measurements (Fig. 1-5-10) was not superior (judged by the R^2) to the model based on plot-level data (Fig. 1-4-7), the former model was based on a larger sample size, $n = 53$, compared to the latter ($n = 26$). Therefore, the individual hardwood tree data extracted from all 55 mined plots, as opposed to the data from 26 hardwood plots (Substudy 4), was regarded as a superior data set for modeling hardwood total tree carbon content on mined land.

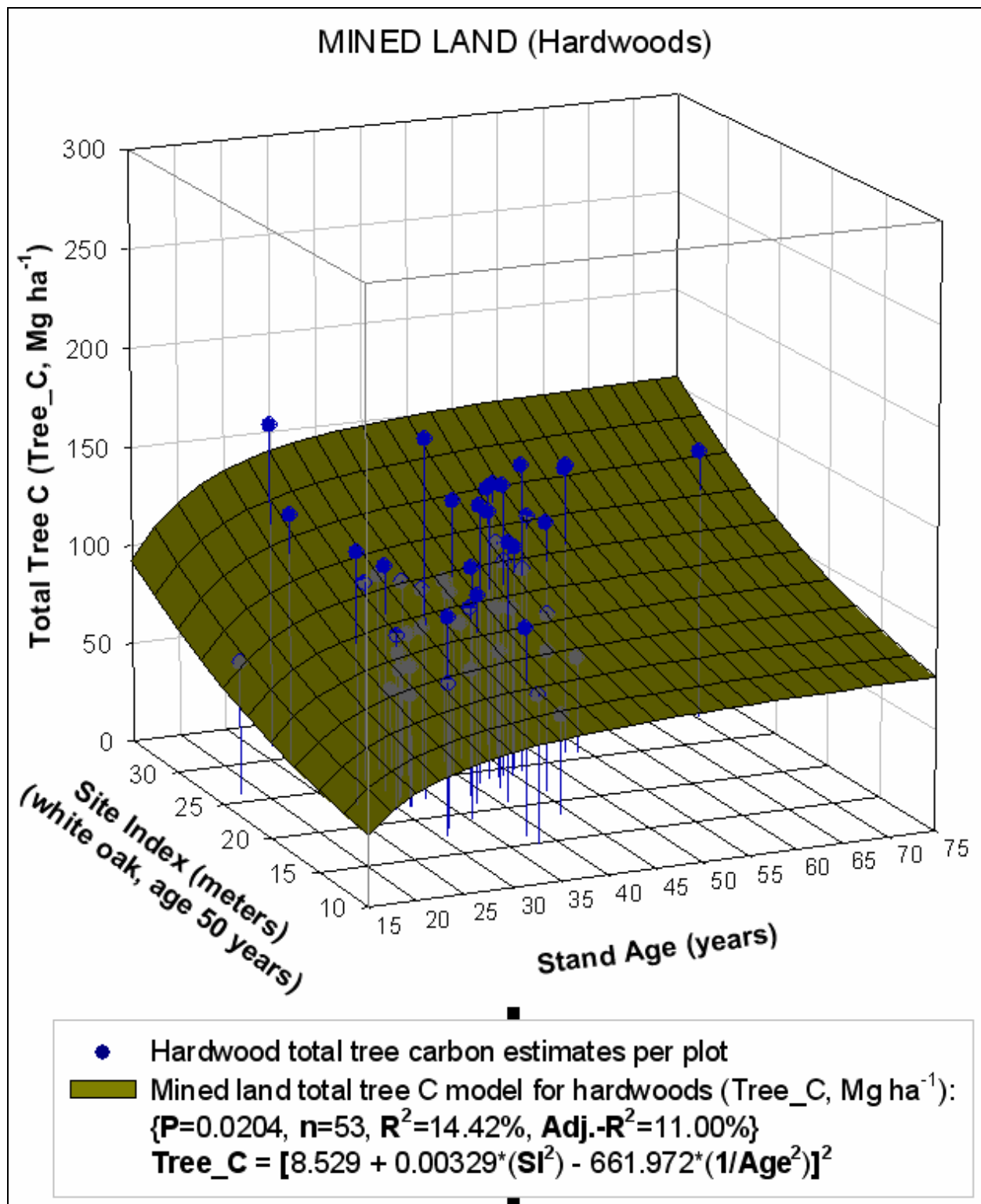


Figure 1-5-10. Total tree carbon content prediction model for hardwood trees for forest stands established on mined lands in the Midwestern and Appalachian coalfields. These data were extracted from a larger data set reported by Rodrigue (2001) and were analyzed using the methods described in the text.

We used identical methods and equations from the literature, similar to the white pine modeling approach above, to estimate the carbon content in tree stems, branches, leaves, and roots (Jenkins et al., 2003). We calculated the expected average tree DBH based on the total tree carbon predictions in Figure 1-5-10 using the reported relationship between aboveground biomass and DBH for mixed hardwoods, *aboveground biomass, kg* = $\text{Exp}(-2.4800 + 2.4835 * \text{Ln}(\text{DBH}, \text{cm}))$ (Jenkins et al., 2003), as shown below:

$$\text{DBH}(\text{cm}) = \text{Exp} \left[\frac{\text{Ln} \left(\frac{\text{TotTreeC, Mg ha}^{-1} * 1000 * 2 / (\text{Trees, ha}^{-1})}{1 + 0.20} \right) + 2.4800}{2.4835} \right]$$

where *TotTreeC, Mg ha⁻¹* = predicted total tree carbon content in Mg ha⁻¹ units; multiplication by (1000*2) is a unit conversion factor to convert carbon content in Mg ha⁻¹ units to biomass content in kg ha⁻¹ units, assuming that 50% of biomass is carbon; *Trees ha⁻¹* = tree density per hectare (Fig. 1-5-9); division by (1+0.20) is based on the assumption that the root biomass for hardwood trees with DBH up to 50 cm is approximately equal to 0.20**aboveground biomass* (derived by Jenkins et al., 2003), hence, *total tree biomass* = *aboveground biomass**(1+0.20).

Using the expected average DBH estimate, we computed the total tree carbon content (Mg ha⁻¹) for future stands that could be established at land owner-specified tree spacing that could differ from the tree density graph in Figure 1-5-9. Total tree carbon content was estimated as *TotTreeC* (Mg ha⁻¹) = (1+0.20)*(($\text{Exp}(-2.4800 + 2.4835 * \text{Ln}(\text{DBH}, \text{cm}))$) / 1000) * (*Tree_density*), where *Tree_density* is the expected number of live trees per hectare when trees are planted at 2.4x3.0 m tree-spacing for a total 1345 trees ha⁻¹ (adopted from Casselman et al., 2006) and predicted mortality rate based on the findings by Harcombe (1987). Carbon content in aboveground biomass of hardwood trees was estimated as (*TotTreeC, Mg ha⁻¹*)*100/120, and carbon content in root biomass was (*TotTreeC, Mg ha⁻¹*)*20/120. We estimated the fraction of the total tree carbon within the remaining tree components, stem, leaves, and branches as follow: stem (bark+wood), Mg C ha⁻¹ = (*TotTreeC, Mg ha⁻¹*)*(100/120)*($\text{Exp}(-2.0129 - 1.6805/\text{DBH}) + \text{Exp}(-0.3065 - 5.4240/\text{DBH})$); leaves, Mg ha⁻¹ = (*TotTreeC, Mg ha⁻¹*)*(100/120)*($\text{Exp}(-4.0813 - 5.8816/\text{DBH})$); and branches (Mg ha⁻¹) were computed as the difference (*TotTreeC, Mg ha⁻¹*)*(100/120) – (*stem, Mg C ha⁻¹*) – (*leaves, Mg C ha⁻¹*). The leaves and root biomass distribution equations were used to model the litter layer and root carbon pools in our DSS. We assumed that the litter layer biomass in hardwood stands would be decomposing at rates similar to the findings by McLaugherty (1985); that is, approximately 29.83% of litter layer biomass (estimated as the average for sugar maple, bigtooth aspen and white oak stands) would remain after 2 years. We modeled root biomass decomposition based on the findings by Silver and Miya (2001), assuming that hardwood tree roots (coarse and fine) would decompose and completely mineralize in approximately 7 years.

Similar to the approach used for white pine stands, we assumed that tree mortality as fraction of live trees within a DBH-size class in undisturbed hardwood forests (Harcombe, 1987) could be used to approximate tree mortality on mined land. We developed a regression equation for DBH classes ranging from 1 to 51 cm based on mortality (dependent variable) and DBH (dependent variable) data for five hardwood species (*Quercus rubra* (northern red oak), *Acer rubrum* (red maple), *A. saccharum* (sugar maple), *Fraxinus americana* (White ash), and *Q. alba*

(white oak)) in Harcombe (1987); the resulting regression equation was as follows: Mortality Rate = $\text{Exp}[-1.62168 + 0.00363 * (\text{DBH,cm})^2 - 0.23074 * (\text{DBH,cm})]$, ($n = 137$, $P < 0.0001$, $R^2 = 48.53$, $\text{Adj.-}R^2 = 47.77$). For trees of DBH-size class outside the 1-51 cm range, we assumed negligent mortality.

We allowed the user to select an appropriate stand management regime from three available options: no thinnings, one thinning (scheduled between stand age 34 and 44 years), and two thinnings (the first being carried out between stand age 34 and 44 years, and the second between stand age 45 and 55 years). For any one of the thinning scenarios, the rotation age could be set between stand age 60 and 70 years. Following each thinning, we assumed 70% increase in tree growth with allowed a range between 0% and 250 %, due to a greater availability of resources and open space from removed trees (assumed 30%) during each thinning operation. We assumed that approximately 15% of harvestable biomass (stems and branches) removed during thinnings would remain on the site and become part of the litter layer carbon pool. We assumed that the harvesting residue from thinning of hardwood stands would have decomposition rates similar to hardwood foliage.

Similar to the modeling approach used for white pines, SOC accumulation under hardwood vegetation was modeled from empirical data reported by Rodrigue (2001) for forested mined sites in the Midwestern and Appalachian coalfields (Substudy 4). The analyses of these data, representing hardwood forests established on reclaimed loose mine spoils, indicated that mean SOC content accumulation to 2-m depth after 60 years could be predicted by site index using the following equation ($P = 0.02$, $n = 26$, $R^2 = 19.79\%$, $\text{Adj.-}R^2 = 16.45\%$): $\text{SOC}_{0-200 \text{ cm}}, \text{Mg ha}^{-1} = [10.887 - 0.239 * (\text{SI})]^2$. This result was based on the assumption of a steady rate of SOC accumulation through the years of stand development. We assumed a linear relationship between SOC content and stand age in order to predict SOC with time. We also assumed that after a certain period of time, SOC content would reach a new equilibrium level, approximately that of undisturbed forests, after which there would be no further increase in soil carbon content. The regression equation used to estimate the equilibrium SOC level was similar to that used for pine stands. We assumed that the MCD of the forested mine soil is 50 cm, meaning that about 85% of the total profile (0-200 cm) SOC content would be measurable (Figure 1-5-7b).

Modeling the Effects of Forest Establishment Practices on Tree Growth and Survival on Mined Land

We used the tree survival and growth results reported by Casselman et al. (2006) to model total tree carbon and SOC accumulation for three levels of intensity of forest establishment practices, ranging from WC, WC + T, WC + T + F for three forest types (Figure 1-5-11). Based on the reported similarities between a siltstone- and a shale-derived mine soil (Casselman et al., 2006), we assumed that tree growth and survival on siltstone-derived mine soils will be comparable to shale-derived mine soils.

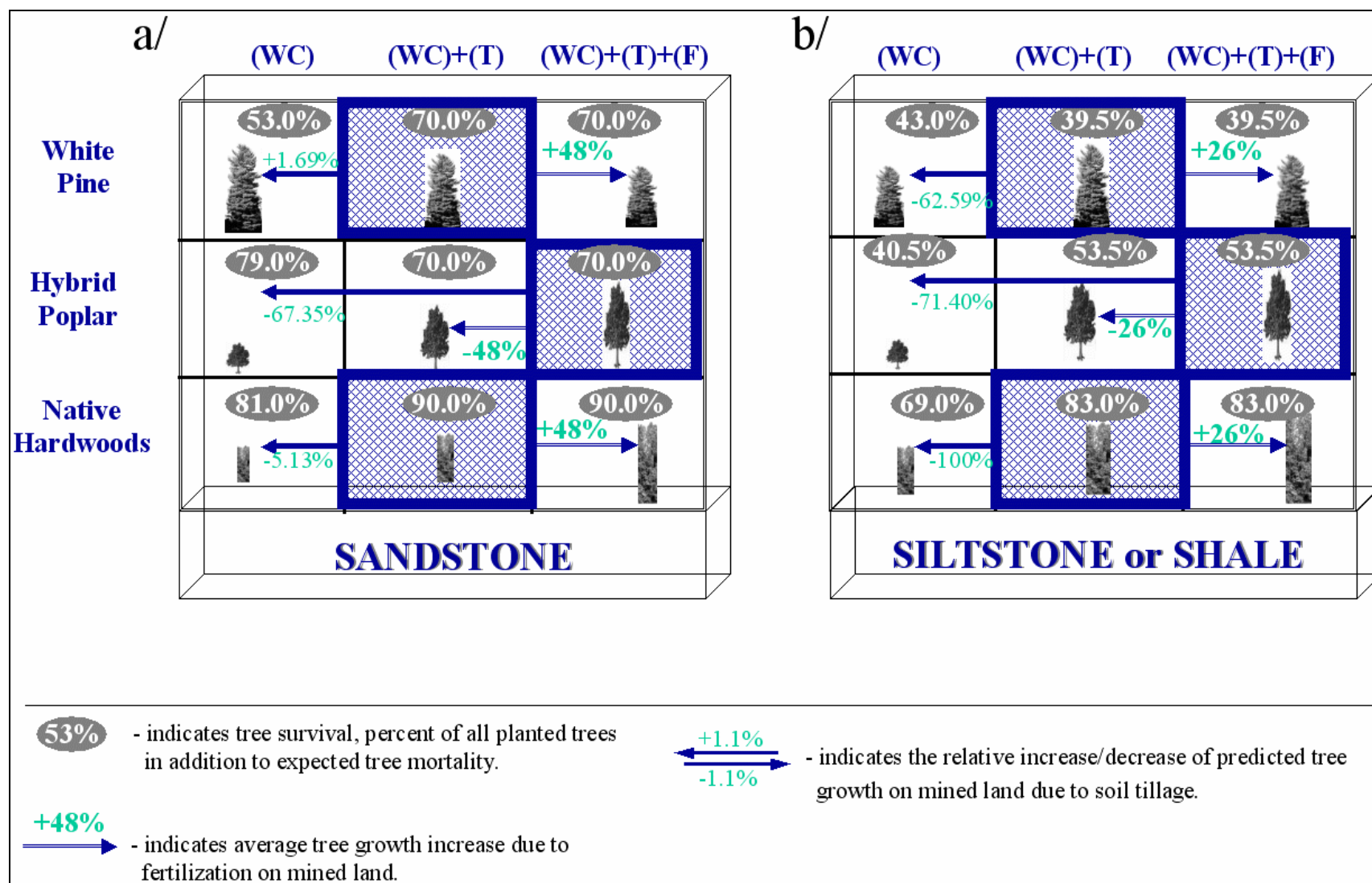


Figure 1-5-11. Modeling the effects of forest establishment practices on the survival and growth of three forest types, white pine, hybrid poplar, and native hardwoods, established on (a/) sandstone- and (b/) siltstone- or shale-derived mine soils based on data by (Casselman et al., 2006). Tree growth increase due to fertilization on sandstone or siltstone (shale) spoils was computed as the average for all species of the respective spoil type.

Based on the finding by Casselman et al. (2006), we developed tree growth and survival correction factors (i.e. relative increase/decrease) due to the effect of tillage and fertilization. The WC treatment was required in all silvicultural treatment intensities to warrant minimal tree survival and, as such, study plots with WC treatment could be considered as controls in this study (Casselman et al., 2006). Because the first growing-season findings by Casselman et al. (2006) agreed well with older studies, we used the relative increase/decrease in tree survival and height growth due to soil tillage reported by Casselman et al. (2006) to modify the total tree carbon models developed from available data for hybrid poplar, white pine, and native hardwoods (discussed in previous sections) (Fig. 1-5-11). Although Casselman et al. (2006) calculated tree volume data, computed as ground stem diameter squared and multiplied by height, because of the young age of the seedlings, we considered that relative differences in tree height growth would better reflect the effects of forest establishment treatments.

Similarly, we developed tree growth correction factors due to fertilization using the average effects (considering both negative and positive effects) for sandstone and siltstone (shale) spoils reported by Casselman et al. (2006). We assumed that there would be negligible change in tree survival due to fertilization, i.e. tree survival would be similar for the WC + T and the WC + T + F treatments. There is a scarcity of experimental data for the effect of fertilization on tree growth and survival on mined land, except for individual reports stating that fertilization could increase tree growth but not survival, and these effects were prominent only during the first year of stand establishment (Rodrigue, 2001).

On one hand, because the hybrid poplar data used to develop the total tree carbon model for our DSS was derived from managed forest stands that had weed control, tillage, and fertilization treatments prior to stand establishment, the resulting model was assumed to best depict tree growth rates on mined land sites of highest management intensity, WC+T+F (Fig. 1-5-11). On the other hand, the white pine and native hardwood total tree carbon models were based on data from unmanaged forest stands growing on loosely dumped mine spoils, implying a tilled-like soil and a weed-free tree growing environment. Therefore, the resulting models for white pine and native hardwood stands were assumed to best match the tree growth rates on mined sites of medium management intensity, i.e. WC+T treated plots (Fig. 1-5-11).

For example, let $Biom_{HP}$ be the hybrid poplar total tree carbon model as a function of SI and stand age, developed from empirical data from hybrid poplar stands established on undisturbed soils that have received WC+T+F treatments prior to stand establishment. Then, the total tree carbon content model for hybrid poplar stands established by the WC+T+F treatment on a sandstone-derived mined site would be $Biom_{HP} * 70/100$, where 70% is the expected initial tree survival on these soils (Fig. 1-5-11). Similarly, the total tree carbon model for hybrid poplar stands established on siltstone- or shale-derived mined sites would be $Biom_{HP} * 53.5/100$. In these examples, multiplying $Biom_{HP}$ by percent tree survival determines the correct number of trees that had survived after one growing-season for the site conditions described, and that would be growing and sequestering carbon at rates similar to those observed on undisturbed lands following WC+T+F forest establishment treatments. These models, as described in previous sections, accounted for the effect of a landowner-selected initial tree density and tree mortality curves reported by Harcombe (1987) on the total tree carbon content; tree density was modeled as a function of tree size (expected average tree DBH).

In order to determine the proper tree growth rate of hybrid poplar stands established on WC and WC+T treated plots, we used the average increase/decrease in tree height growth relative to

tree height growth on all WC+T+F experimental plots (Fig. 1-5-11). This was necessary because there was insufficient data in the literature to model hybrid poplar tree growth on WC and WC+T treated mined lands. For example, the hybrid poplar height growth on a sandstone-derived mine soil was 48% (estimated as the average for all species) and 67.35% lower on the WC+T and WC experimental plots, respectively, relative to the height growth observed on the WC+T+F experimental plots (Fig. 1-5-11). Therefore, the respective total tree carbon prediction models for the WC and WC+T treated mined lands would be $(79/100) * Biom_{HP} * (1-67.35/100)$ and $(70/100) * Biom_{HP} * (1-48/100)$, where 79 and 70% are the respective tree survival rates and $(1-67.35/100)$ and $(1-48/100)$ are the respective tree height growth adjustment factors.

We used a similar approach to develop total tree carbon models for the treatment scenarios presented in Figure 1-5-11 for white pine and native hardwood stands established on mined land. For example, if $Biom_{HW}$ was the total tree carbon model for forest stands established by planting trees on loosely dumped mine spoil, then the total tree carbon model for stands established by the WC+T treatments on sandstone- and siltstone/shale-derived mine lands would be $(90/100) * Biom_{HW}$ and $(83/100) * Biom_{HW}$, respectively (Figure 1-5-11). For forest stands established on sandstone-derived mined lands by the highest treatment intensity, WC+T+F, the total tree carbon model would be $(90/100) * Biom_{HW} * (1+48/100)$. Multiplying $Biom_{HW}$ by percent tree survival assured the correct number of growing trees, and a second multiplication by the relative tree growth adjustment factor, estimated as $[1 +/- (1/100) * \text{percent increase/ decrease of tree height growth}]$, assured estimation of the correct tree growth rate across treatment intensities.

Using an identical approach, similar to the total tree carbon models described above, we developed SOC models for mined land for all experimental scenarios shown in Figure 1-5-11. Total tree carbon and SOC models for the tree species groups and for the treatment intensities depicted in Figure 1-5-11 were implemented in our DSS. In the future, as more data become available, especially describing the effect of fertilization on tree growth and survival as a function of stand age, we will develop new and updated total tree carbon and SOC models, which could be implemented in future iterations of the DSS.

Modeling Carbon Storage in Wood Products and Carbon Emissions from Forest Establishment Practices, Tree Harvesting, Biomass Transportation, Wood Product Manufacturing, and Biomass Burning

Based on reports in the literature regarding fossil fuel usage and GHG emissions in the atmosphere from various operations in the forest industry, starting from site preparation (tillage and fertilization) to wood products decomposition in landfills, we estimated the amount of carbon in the two major post-tree-harvesting carbon pools. The first carbon pool depicted the amount of carbon that was permanently sequestered in landfills, and the second was the carbon pool of GHGs emitted in the atmosphere from biomass burning and fossil fuel usage. Because fluxes in both pools exist together, we modeled the simultaneous carbon emission and carbon storage at each step along the carbon cycle within the framework of forest industry operations (Fig. 1-5-12).

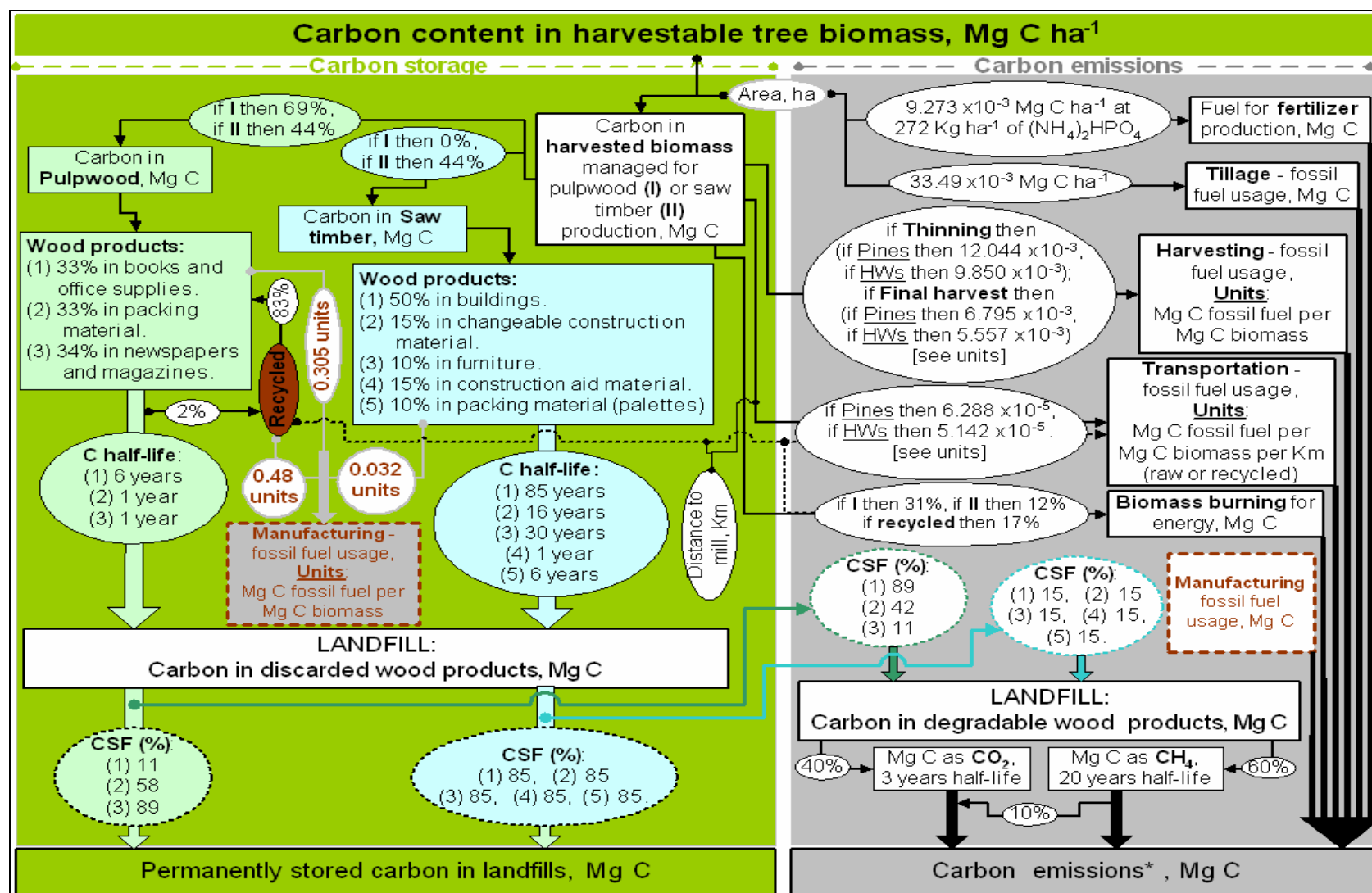


Figure 1-5-12. Modeled tree biomass carbon storage and flow, and carbon emissions from fossil fuel usage and biomass burning for energy, as a result of forest industry operations. Percent CSF indicates carbon storage factor for a specific wood product; (* indicates the amount of elemental carbon, in CO₂ form, which represents the cumulative greenhouse warming potential of all GHGs emitted from fossil fuel usage and biomass burning (after Karjalainen and Asikainen, 1996)).

We assumed that when fertilization was recommended for forest establishment on mined sites, the default fertilizer application rate would be 272 kg ha^{-1} of diammonium phosphate (adopted from Casselman et al., 2006), for the production of which approximately $9.273 \times 10^{-3} \text{ Mg C}$ would be released in the atmosphere from fossil fuel burning (Schlesinger, 1999) (Fig. 1-5-12). In our DSS, we allowed the option for the user to enter a modified fertilizer amount, as well as to specify the frequency of fertilizer application.

Assuming that planted forests on mined land would be managed with a goal to maximize wood production and carbon sequestration, it could be expected that these stands would be fertilized yearly (adopted from Coleman et al., 2006), as opposed to fertilization at planting and tree closure when the demand for nutrient is highest. Similar to the fertilization treatment, we assumed that certain mined sites would require tillage treatments, which would result in carbon emissions of approximately $33.49 \times 10^{-3} \text{ Mg C ha}^{-1}$ (Karjalainen and Asikainen, 1996) (Fig. 1-5-12). We assumed that, when tillage was recommended, the mined site would be tilled at each forest establishment event, following tree harvesting from the previous rotation. Due to lack of data in the literature, we assumed negligible amounts of carbon emissions from fossil fuel usage for herbicide production and application on mined land.

We assumed that established forest stands, except the short-rotation hybrid poplar, could be thinned at least once prior to final tree harvesting. We adopted the baseline carbon density estimates for pine and hardwoods, as well as the fossil fuel usage rates during certain forest operations reported in Karjalainen and Asikainen (1996) to account for the amount of GHGs emitted in the atmosphere during tree harvesting, haulage and transportation of pine and hardwood biomass (Fig. 1-5-12). We assumed that during thinning of pine and hardwood stands, 12.0444×10^{-3} and $9.8501 \times 10^{-3} \text{ Mg fuel C per Mg biomass C}$, respectively, would be emitted into the atmosphere (adopted from Karjalainen and Asikainen, 1996). These estimates represented the cumulative global warming potential of all emitted GHGs, including CO_2 , N_2O , CO , CH_4 , NO_x , and NMVOC, expressed as the GWP of CO_2 . During final harvest of pine and hardwood stands, we assumed that 6.7950×10^{-3} and $5.5570 \times 10^{-3} \text{ Mg fuel C per Mg biomass C}$, respectively, would be emitted in the atmosphere (Karjalainen and Asikainen, 1996) (Fig. 1-5-12). Following thinning or final harvest, we assumed that 6.2877×10^{-5} and $5.1422 \times 10^{-5} \text{ Mg fuel C per Mg biomass C km}^{-1}$ would be emitted in the atmosphere during loading, transportation, and unloading of pine and hardwood biomass, respectively (Fig. 1-5-12).

In the post-tree-harvesting section of the carbon modeling in our DSS, we estimated the carbon storage in wood products assuming that the entire yearly yield of roundwood would be transformed into wood products within a one-year period, without any additional delays. When the forests were managed for pulpwood production, the roundwood would be used to manufacture three types of wood products (Liski et al., 2001): books and office supplies, packing material, and newspapers and magazines. Approximately one-third of the pulpwood would be used to manufacture each of the three types of wood products (Figure 1-5-12).

The energy (e.g., electricity) used for wood product manufacturing would be obtained from two sources. The first source would be fossil fuel burning, estimated at $0.305 \text{ Mg of fuel C per Mg of biomass C}$, and the second source would be the burning of the nonprocessable tree biomass (approximately 31%) (Fig. 1-5-12).

We assumed that approximately 2 % of pulpwood-based wood products would be recycled (derived from Liski et al., 2001) (Fig. 1-5-12). In our DSS, we provided the option for the user

to enter an appropriate recycling intensity as percent of total paper and pulp production that would best match the practices of a certain region or a certain country. Based on the estimates by Liski et al. (2001), we assumed that 83% of the recycled biomass would be used to manufacture more pulp and paper products, and the rest (17%), ranked as nonprocessable residue, would be burned for energy. Additional energy to process recycled material into products would be used from fossil fuel burning at a rate of 0.48 Mg of fuel C per Mg of recycled C. Calculated as the average of the GHG emissions released during transportation of pine and hardwood biomass, we assumed that 5.7150×10^{-5} Mg fuel C per Mg biomass C per km distance would be emitted in the atmosphere during loading, long-distance transportation, and unloading of recycled wood products. For modeling purposes, we assumed that the average long-haul transportation distance for raw and recycled material would be the same; default distance was assumed to be 100 km, with the option for the user to modify it.

We assumed that the half-life of carbon in pulpwood- and sawtimber-derived wood products were as follow (Liski et al., 2001; Skog and Nicholson, 1998): 6 years for books and office supplies, 1 year for packing material and for newspapers and magazines; 85 years for residential buildings, 16 years for changeable construction material, 30 years for furniture, 1 year for construction aid material, and 6 years for palettes (Fig. 1-5-12). We adopted the first-order decay curve, described in Miner (2003), to model the amount of carbon in wood products in use and the amount of carbon in wood products in landfills.

We used the carbon storage factor estimates reported in Miner (2003) to model the amount of permanently stored carbon in landfills, in the form of decay-resistant wood biomass fragments, as well as to model the carbon amount in degradable wood products in landfills (Fig. 1-5-12). We assumed that, following their time in use, 11% of the carbon in discarded books and office supplies would remain in landfills permanently, while 89% would be decomposed to CO₂ and CH₄. Similarly, 58% of the carbon in packing material and 89% of the carbon in newspapers and magazines would be stored permanently in landfills, while the remainder of each type of wood products would be decomposed to CO₂ and CH₄ (Fig. 1-5-12). For each of the five types of wood products manufactured from sawtimber, we assumed that 85% would be permanently stored in landfills, and 15% would be decomposed to CO₂ and CH₄ (Miner, 2003) (Fig. 1-5-12).

Under the capping soil layer of landfills, we assumed that 40% of the carbon in degradable wood products would be decomposed to CO₂ and 60% would be decomposed to CH₄. We assumed that CO₂ and CH₄ would slowly be released in the atmosphere in a fashion that could be described with a first order decay function (adopted from Skog and Nicholson, 1998). We also assumed that the half-life of the CO₂-carbon was 3 years, and was 20 years for CH₄-carbon so that 10% of the methane would be oxidized to carbon dioxide before moving out of the landfill (adopted from Micales and Skog, 1997) (Fig. 1-5-12).

Model Simulation Scenarios

We estimated the net carbon sequestration from forests established on mined land for 60 years into the future. For each forest type and silvicultural treatment, we predicted the net carbon sequestration in forests on mined land using the DSS developed in this study. We contrasted and compared the net carbon stocks accumulated under (1) hybrid poplar forest vegetation managed at a 15-year rotation for a total of four harvests at year 60 of the carbon project, (2) white pine forest vegetation with two final harvesting events at year 60, scheduled 30 years apart, and (3) hardwood forest vegetation harvested at the end of a 60-year project. For

each simulation scenario, the net carbon stock was estimated as the difference between gross carbon stock sequestration in the forest ecosystem and wood products (including sequestration in landfills) and carbon emissions (including carbon emissions from landfills). We compared the results of all analyses in order to identify the best forest establishment practice that would lead to maximizing carbon sequestration on mined land.

We also performed sensitivity analyses to estimate the relative change in predicted net carbon sequestration associated with changes in the magnitude of selected input variables depicted in Figure 1-5-4, such as site index, rotation age, thinning age (if thinning option was selected), harvesting regime, and planting density. Information about the relative effect of each input variable on the net carbon predictions made by the DSS could be used to select the most appropriate forest establishment and management treatments that would maximize carbon sequestration on mined land.

Results and Discussion

Forest Establishment and Management Scenarios

The predicted amount of net carbon stock sequestered in forest ecosystems (including the carbon in wood products in use and in landfills) established on mined lands for 60 years ranged from 20 to 235 Mg C ha⁻¹ (Fig. 1-5-13). In general, the carbon stocks were higher on sandstone-derived mine soils compared to siltstone/shale-derived soils for each of the three forest type groups. Higher net carbon stocks for the three forest types on sandstone-derived soils were directly related to higher tree survival compared to siltstone/shale-derived soils (Fig. 1-5-11). Because of high compaction levels on siltstone/shale-derived mine soils, tree growth in hardwood stands established by the WC treatment could be severely retarded; this was depicted by a 100% reduction in tree growth when no tillage was applied on siltstone/shale spoil type (Fig. 1-5-11). As a result, we did not estimate the net carbon content in hardwood stands established by the WC treatment on sandstone/shale-derived mine soils (Fig. 1-5-13).

The greatest differences between the two spoil types were observed for white pine and mixed hardwood stands (Fig. 1-5-13). The net carbon stock in white pine stands established by the WC treatment on sandstone spoils (106 Mg C ha⁻¹) was about 238% greater compared to a forest stand established by the same treatment on siltstone/shale spoils. The lowest differences between the two spoil types were observed in hybrid poplar stands (Fig. 1-5-13).

The highest net amount of carbon sequestration (at rotation age 60) by forest establishment on mined land was 234.6 Mg C ha⁻¹ in mixed hardwood stands established on sandstone spoils by the highest intensity treatment (WC+T+F). The second highest net carbon content (184.2 Mg ha⁻¹) was also sequestered in mixed hardwood forests established by the WC+T+F treatment on siltstone/shale spoil (Fig. 1-5-13). The third highest net carbon stock (ranging between 135 and 160 Mg ha⁻¹) was sequestered in mixed hardwood stands established by either WC or WC+T treatments on SS spoil or by WC+T treatment on SiS spoil, and in white pine stands established by the WC+T+F treatment on sandstone spoils (Fig. 1-5-13).

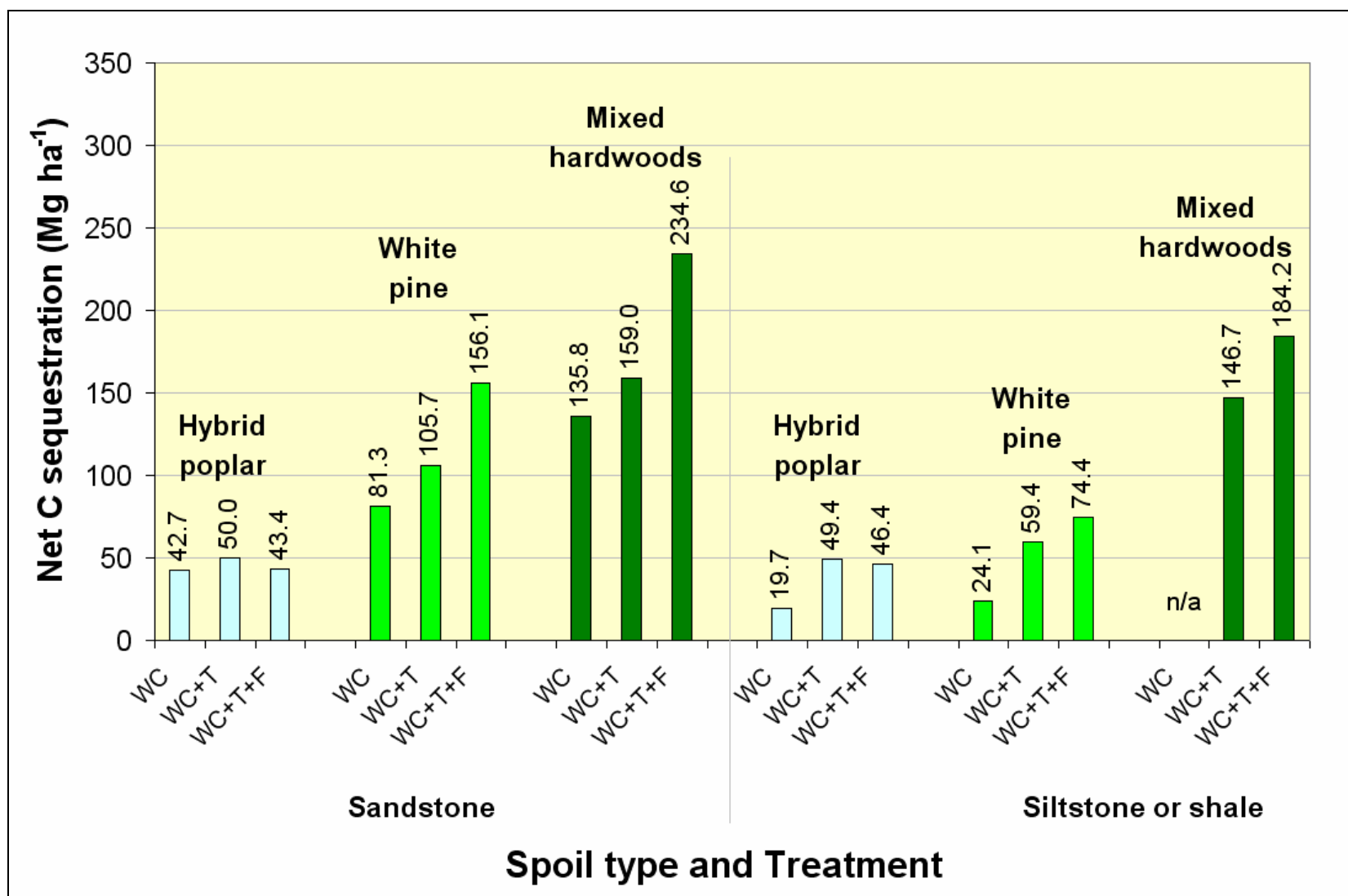


Figure 1-5-13. Net carbon sequestration after 60 years in forest ecosystems of three forest type groups (hybrid poplar stands with rotation age 15 years, white pine stands with rotation age 30 years, and mixed hardwood stands with rotation age 60 years) established on mined lands of two spoil types (sandstone and siltstone/shale) following three intensities of forest establishment treatments (weed control (WC), WC plus tillage (WC+T), and WC+T plus fertilization (WC+T+F)). The site index was assumed to be 25 m.

These results were counter-intuitive at first because despite the relatively faster growth of hybrid poplar and white pine trees compared to mixed hardwood species such as oaks, ash and hickory, the net carbon amounts sequestered in vegetation biomass, the soil, and wood products from hybrid poplar and white pine stands were lower. Usually, faster growing tree species would accumulate more cumulative carbon in vegetation biomass compared to species of slower growth rates. However, soon after harvesting, it should also be expected that some fraction of the carbon in the vegetation biomass would be emitted back to the atmosphere.

Because the DSS was designed to estimate the amount of carbon emissions from wood product manufacturing, burning, and biomass decomposition in landfills, the larger amounts of sequestered carbon in fast growing trees were cancelled out by proportionally larger amounts of emitted carbon from the above-mentioned sources, especially from the fraction of vegetation biomass used for pulp manufacturing. This fact was clearly shown in the results for hybrid poplar stands managed at 15-year rotations (Fig. 1-5-14).

After two rotations (30 years), about 45 Mg C ha^{-1} that was sequestered in the forest ecosystem and wood products was matched by carbon emissions back to the atmosphere equivalent to 8 Mg C ha^{-1} estimated for the land mass of the originating forest (Fig. 1-5-14). After four rotations (60 years), the amount of sequestered carbon in hybrid poplar stands (about 82 Mg C ha^{-1}) was matched by 32 Mg C ha^{-1} carbon emissions to the atmosphere.

Clearly, there is a tendency that the longer the time that sequestered carbon in tree biomass (above- and below-ground) stays locked in the forest ecosystem, the greater the amount of net carbon sequestration from this forest in the long-term. At the end of a 120-year period, the net carbon sequestration from white pine stands managed at 30-year rotations was one order of magnitude greater than hybrid poplar stands. The most significant differences between the carbon pools in hybrid poplar and white pine stands were the soil, above-ground biomass, and wood products in use. The cumulative carbon content in the soil, above-ground, and wood products in use from hybrid poplar stands represented 59, 10, and 0.5%, respectively, of the total carbon content (94 Mg C ha^{-1}) sequestered 120 years after project initiation (Fig. 1-5-14). In contrast, the cumulative carbon content in the soil, above-ground, and wood products in use from white pine stands represented 8, 32, and 12%, respectively, from the total carbon content at year 120, which was 398 Mg C ha^{-1} (Fig. 1-5-15).

Compared to the carbon emissions from hybrid poplar and white pine biomass decomposition and burning described above, there were negligible carbon emissions (mostly from fossil fuel usage for soil tillage and fertilizer production) from the mixed hardwood stands (Fig. 1-5-16). At rotation age 60 years, the hardwood stands sequestered approximately 235 Mg C ha^{-1} , while approximately 0.6 Mg C ha^{-1} was emitted to the atmosphere until then from soil tillage and fertilization treatments. About 71% of the net sequestered carbon was in above-ground biomass, followed by the carbon stock in root biomass (14%), soil (11%), and litter layer biomass (4%) (Fig. 1-5-16).

The results in Figure 1-5-13 could be used to make informed decisions as to what forest establishment practice and forest type group may be chosen for a carbon sequestration project. Because of the potential to sequester 50% more net ecosystem carbon at the end of a 60-year rotation, it was logical to consider the WC+T+F treatment over the WC+T to establish mixed hardwood stands on mined land.

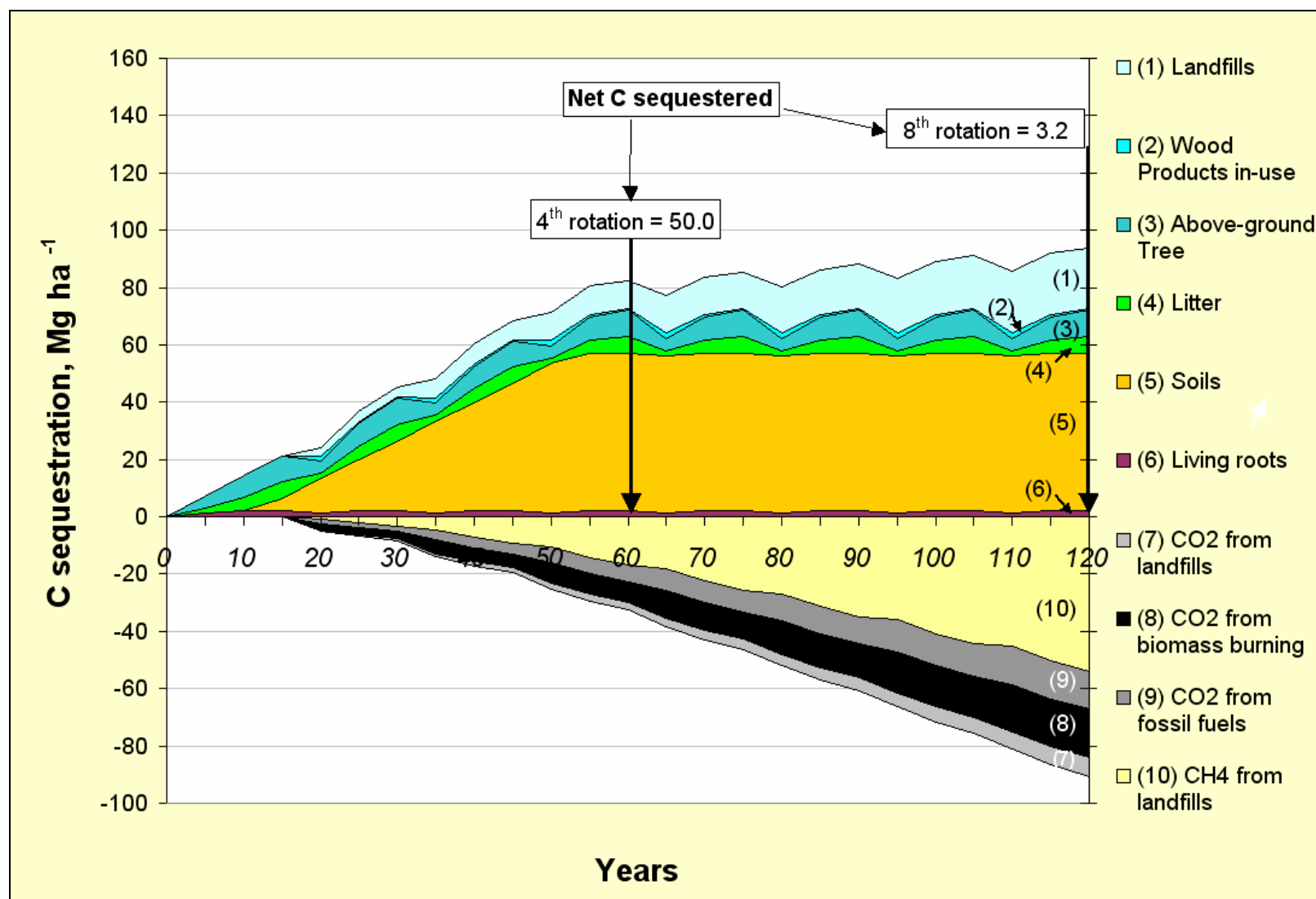


Figure 1-5-14. Cumulative carbon sequestration in the forest ecosystem components after eight 15-year rotations in hybrid poplar stands established on sandstone-derived spoil (SI = 25m) following weed control and soil tillage applications. Negative values indicate carbon emissions from decomposition of discarded vegetation biomass in landfills, or biomass and fossil fuel burning for energy.

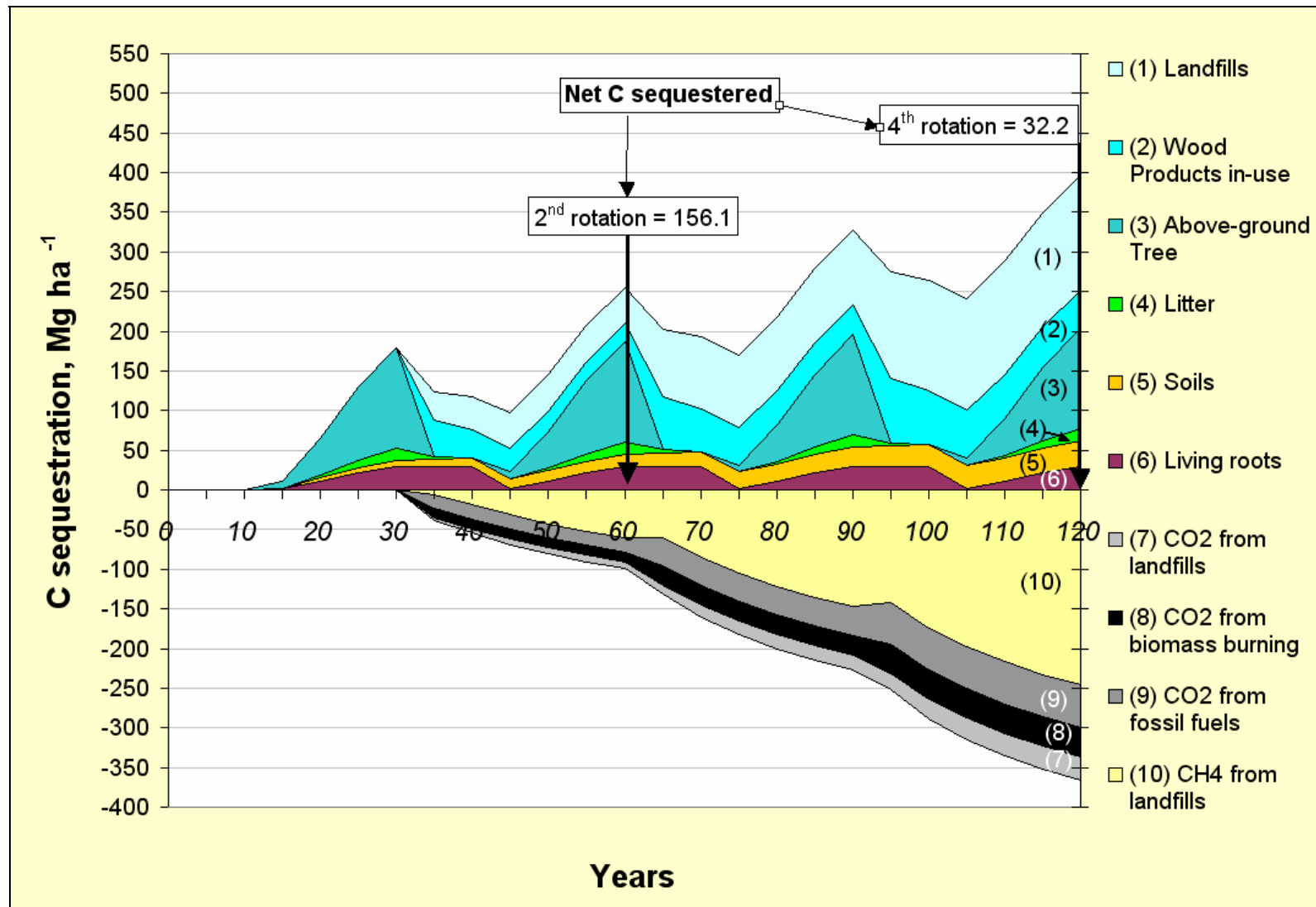


Figure 1-5-15. Cumulative carbon sequestration in the forest ecosystem components after four 30-year rotations in white pine stands established on sandstone-derived spoil (SI = 25m) following weed control, soil tillage, and fertilizer applications. Negative values indicate carbon emissions from decomposition of discarded vegetation biomass in landfills, or biomass and fossil fuel burning for energy.

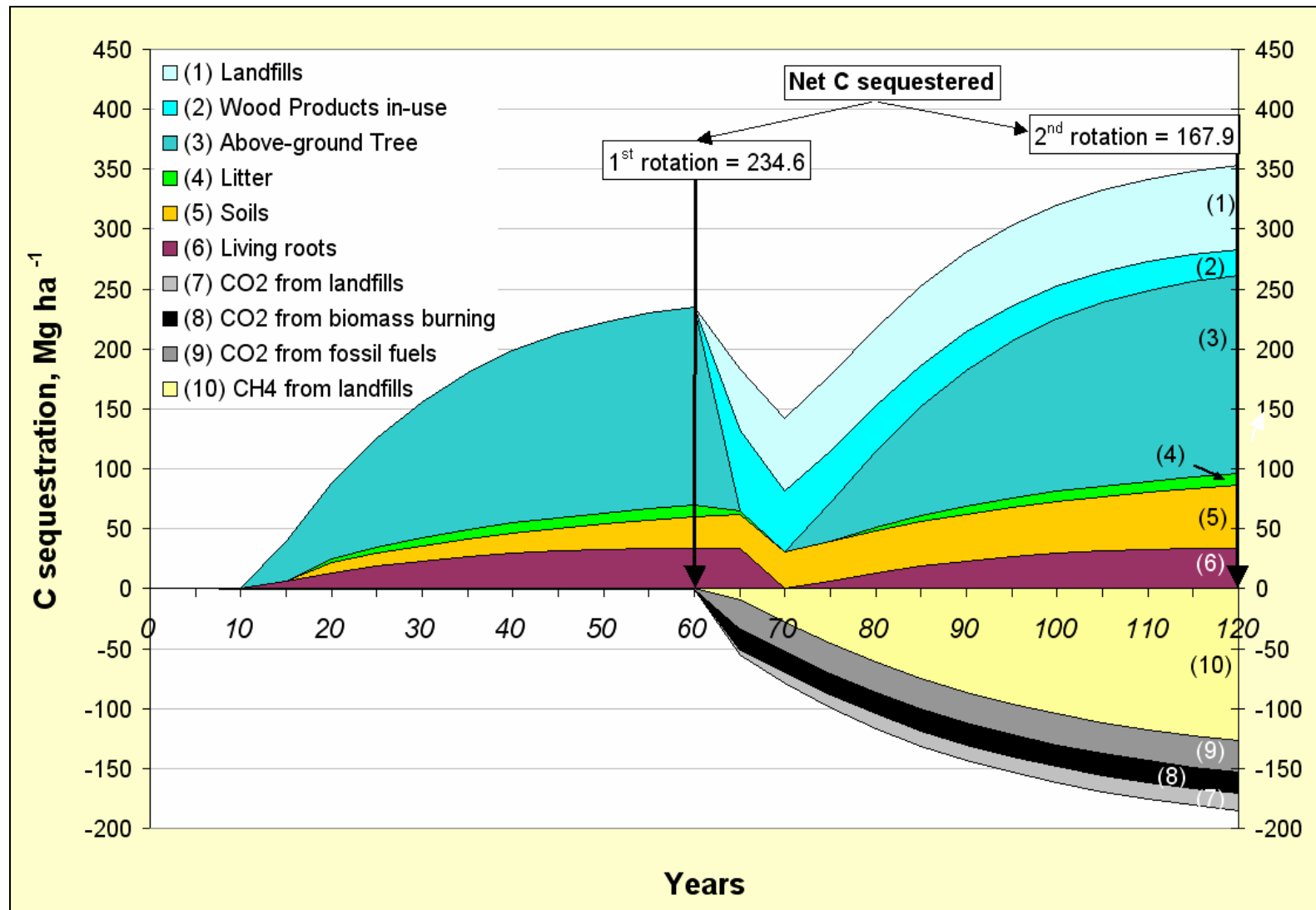


Figure 1-5-16. Cumulative carbon sequestration in the forest ecosystem components after two 60-year rotations in mixed hardwood stands established on sandstone-derived spoil (SI = 25m) following weed control, soil tillage, and fertilizer applications. Negative values indicate carbon emissions from decomposition of discarded vegetation biomass in landfills, or biomass and fossil fuel burning for energy.

Immediately after tree harvesting at rotation age 15, 30, and 60 years, carbon emissions started from fossil fuel burning for energy and biomass decomposition from landfills in hybrid poplar, white pine, and hardwood stands, respectively (Figs. 1-5-14, 1-5-15, 1-5-16). At the end of the second rotation in hardwood stands, 120 years after forest establishment, the cumulative estimate for carbon emissions to the atmosphere as CO₂ and CH₄ was equivalent to 185 Mg C ha⁻¹ estimated for the land mass of the originating forest (Fig. 1-5-16). In contrast, for the same time period, due to harvesting at an earlier time, the cumulative carbon emissions from white pine stands were 365 Mg C ha⁻¹, twice the amount from hardwood stands (Fig. 1-5-15). In both cases, the largest component of the cumulative carbon emissions (nearly 70%) was CH₄, followed by carbon from fossil fuel burning for energy (15%), biomass burning for energy (10%), and the remainder was emitted as CO₂ from landfills (Figs. 1-5-14, 1-5-15, 1-5-16).

Although carbon was sequestered in plant leaves in the form of atmospheric CO₂, at the end of its biogeochemical cycle, a fraction of this carbon is emitted back to the atmosphere in a different form, CH₄. Because CH₄ has 21 times the GWP of CO₂, each carbon atom that was sequestered as CO₂ and emitted as CH₄ would negate the sequestration effect of 20 other carbon atoms that were emitted back to the atmosphere (at the end of their biogeochemical cycles) as CO₂ molecules.

Despite the significant effect of CH₄ emissions from landfills, the total carbon emissions from hardwood stands were counteracted by the carbon stock sequestered at 353 Mg C ha⁻¹ in the above-ground tree biomass (47%), landfills (20%), soils (15%), root biomass (10%), wood products in use (6%), and litter layer biomass (3%) (Figure 1-5-16). As a result, the net carbon sequestration at the end of two 60-year rotations in mixed hardwood stands established on sandstone spoil by the WC+T+F treatment was approximately 168 Mg C ha⁻¹ (Fig. 1-5-16). This estimate was computed under the assumption that the WC+T+F silvicultural treatment would be applied prior to tree planting at years 0 and 60 of the project.

It should be noted that carbon estimates for root biomass produced by the DSS were intended to depict the carbon content in live roots (Fig. 1-5-16). A fraction of the carbon content in tree roots is incorporated in the soil following biomass decomposition, while the rest is emitted to the atmosphere as CO₂. While the transfer of carbon from decaying roots to the soil was not exclusively modeled in the DSS, it was accounted for in the SOC models. The SOC models used in the DSS represented the net accumulation of SOC in mine soils due to leaf litter and root biomass decomposition by soil microorganisms.

Due to the new techniques for constructing landfills by using capping soil material, which ultimately inhibits biomass decomposition, in the long-term, the amount of sequestered carbon in landfills could be significantly greater than the carbon stocks in the soil of the originating forest ecosystem. Assuming that landfill constructions are designed to last for more than 100-200 years, the landfill carbon pool would be the second largest carbon pool (following above-ground vegetation biomass) resulting from forest establishment on mined lands. The results showed that after two 60-year rotations, nearly 30% as much carbon was sequestered in landfills compared to the soil carbon pool (Fig. 1-5-16). Similarly, for the same period of time, the carbon stock accumulated in landfills was nearly half the carbon stock in the above-ground vegetation biomass that would be harvested at the end of the second 60-rotation of mixed hardwood forests (Fig. 1-5-16).

These results suggested that the landfill carbon pool could be used as an efficient means to sequester atmospheric CO₂ at rates much greater than the soil carbon pool. An interesting aspect of the landfill carbon pool is that the sequestration of carbon in landfills does not interfere with societal needs and demands for wood products. On the contrary, after their expected time in use, most wood products such as newsprint, furniture, and building materials are placed in landfills as a service to the community and, at the same time, this process would sequester a significant fraction of the atmospheric CO₂.

Sensitivity Analysis

The results from the sensitivity analysis indicated that changes in site index would lead to dramatic changes in the net amount of sequestered carbon in mixed hardwood stands established on sandstone-derived mine soils (Table 1-5-1). For every 1-m increase in site index beyond SI 25 m, the net carbon sequestration would increase by 2.9 and 5.8% after the first and second 60-year rotation, respectively, following forest establishment. Similarly, for lower quality sites (SI < 25 m), for each 1-m decrease in site index below 25 m, the net carbon sequestration at the first and second rotations would decrease by about 2 and 3%, respectively (Table 1-5-1). These results were in agreement with the recommendations of some mined land researchers that reclaiming surface coal-mined lands to higher site quality would significantly increase the carbon sequestration potential of mined sites throughout the Appalachian coalfield.

Table V-1. Sensitivity analysis results for net carbon stock change from hardwood stands due to changes in site index, rotation age, thinning age, harvesting regime, and tree planting density. The information in parentheses relates to the default input of the respective variables. For example, for site index, *height WO50* indicates that site index of white oak at 50 years base age was used in this analysis; *no thin, H @ 60* indicates no thinning and harvest at the end of a 60-year rotation; *thin @ 40* indicates thinning at stand age 40 years; 1345 trees ha⁻¹ was the default tree density. The harvesting regime analysis for +1 *thin* input in the DSS tool indicates only one thinning at age 40 followed by *H @ 60*, while the +2 *thins* input indicates two thinning events at stand age 40 and 50 followed by *H @ 60*.

Variable	Input Change	Net C Stock Change at Rotation	
		1 st Rotation	2 nd Rotation
		----- % -----	
Site Index	+ 10 m	28.8	5.8
(height WO50 = 25 m)	-10 m	-19.0	-29.1
Rotation Age	+ 10 yrs	2.1	-3.6
(no thin, H@60)	- 10 yrs	-5.3	-1.0
Thinning Age	+ 5 yrs	-2.7	-14.8
(thin@40, H@60)	- 5 yrs	3.3	18.3
Harvesting Regime	+ 1 thin	135.15	4.75
(no thin, H@60)	+ 2 thins	-71.89	-301.69
Planting Density	+ 10%	8.9	6.9
(1345 trees ha ⁻¹)	- 10%	-8.9	-6.9

Choosing an appropriate harvesting regime could also increase the net carbon sequestration in mixed hardwood stands established on mined lands (Table 1-5-1). Compared to a no-thinning management scenario, thinning at age 40 followed by a final harvest at age 60 would result in approximately 135 and 5% increase in net carbon sequestration at the end of the first and second 60-year rotation, respectively (Table 1-5-1). This was expected because the modeling procedures in the DSS accounted for (default settings) 70% increase in tree growth that is expected after thinning (removing 30% of all live trees). Similar procedures were used in the DSS to modify tree growth following the second thinning at age 50, when two thinnings were selected. Compared to a no-thinning management scenario, two thinnings, at age 40 and 50, followed by a final harvest at age 60 would result in 72 and 302% decrease in net carbon sequestration measured after the first and second rotations, respectively (Table 1-5-1). The significant decrease in net carbon stock measured at the end of the second 60-year rotation was due to a 20%-increase in carbon emissions (for a total of 1,100 Mg ha⁻¹) from biomass burning and decomposition, which was matched by approximately 30% less carbon stock in the forest ecosystem, wood products in use and in landfills. These results indicated that one thinning (age 40) is desirable in mixed hardwood forests, while a second thinning is not warranted in forest ecosystems established for carbon sequestration.

The sensitivity analysis results for thinning age (assuming only one thinning would be applied at age between 35 and 45, followed by a final harvest at age 60) suggested that thinning at age 35 would increase the net carbon sequestration by 3.3 and 18.3% at the end of the first and second 60-year rotations, respectively, compared to choosing a thinning at age 40 (Table 1-5-1). Once more, due to the relative increase in tree growth due to a greater availability of light, water, and nutrients, thinning a forest stand to an appropriate tree stoking at age 35, as opposed to 5 years later, would result in a significant increase in net carbon sequestration.

The sensitivity analysis for rotation age indicated that delaying the final harvest from age 60 to 70 would result in a 2.1% increase in net carbon stock at the end of the first 70-year rotation, while there would be a 3.6% decrease at the end of the second 70-year rotation (140 years following forest establishment) (Table 1-5-1). However, timber harvest from mixed hardwood stands at shorter rotations (age 50) would result in a 5% and 1% decrease in net carbon stock at the end of the first and second 50-year rotation, respectively (Table 1-5-1). These results indicate that the sooner the vegetation biomass is harvested for wood product manufacturing, the sooner carbon emissions to the atmosphere will start. Therefore, in order to maximize the net carbon sequestration in forests and forest products, longer rotations should be selected that would delay carbon emissions from biomass burning and decomposition.

The net carbon stock predictions for forest establishment on mined land made by the DSS were similar to the findings by other researchers (Liski et al., 2001; Spinney et al., 2005). Liski et al. (2001) concluded that long rotation lengths for Scots pine and Norway spruce stand management were most favorable for accumulating larger carbon stocks with smallest fossil fuel usage. However, Liski et al. (2001) warned that, despite the resulting increase in carbon stocks, long rotation lengths could be less desirable for landowners because of the delay in revenues from timber production.

Based on the results from the sensitivity analysis in this study, the best forest management scenario for mixed hardwood stands on mined lands would be to schedule a thinning at age 35 and a final harvest at age 60 years. Based on mined site conditions, tree species, and the climate, the expected increase in tree growth after stand thinning could be significantly less than the 70%

estimate used in the DSS. Prior to selecting the appropriate management scenario, mined land owners should seek ancillary information about the percent tree removal required on their property to achieve a desired increase in tree growth of selected tree species. Removing trees of poor quality during thinning and providing sufficient resources to selected high-quality timber species such as northern red oak would bring larger revenues at harvest.

The net carbon stocks from forest establishment could be increased further if the initial tree stocking was at least 1345 trees ha⁻¹. For a 10% increase in tree stocking there would be 8.9% and 6.9% increases in net carbon sequestration at the end of the first and second 60-year rotations, respectively (Table 1-5-1). Due to the lower tree survival (i.e., fewer live trees at the end of the rotation) on mined lands compared to undisturbed forests, choosing lower planting density (less than 1345 tree ha⁻¹) could result in a dramatic decrease in the net carbon sequestration potential of forests on mined land (Table 1-5-1). Conversely, planting a sufficient number of tree seedlings on mined land would warrant the establishment of a healthy forest ecosystem that would provide numerous environmental and societal benefits, as well as sequester significant amounts of atmospheric CO₂.

Direct comparison of these results to other models was not possible because these data were based on simulations for non-mined land (Johnsen et al., 2001; Niu and Duiker, 2006; Smith et al., 2006). Modeling assumptions in some studies were very different than relevant results found in others. For example, Johnsen et al. (2001) and Niu and Duiker (2006) assumed that 30% of harvested biomass would remain in wood products 30 years after harvest, while others estimated (based on different assumptions) that 24 % of the carbon in harvestable tree biomass from the first rotation would remain sequestered in wood products in use and in landfills 30 years after the harvest (derived from Smith et al., 2006). The results from our DSS for white pine and hardwood stands indicated that approximately 20% of the harvestable biomass would be in wood products in use 30 years after harvest. Although the analyses of our DSS may provide useful qualitative information about the effects of different forest establishment practices on the carbon sequestration potential of mined land, the DSS uses many gross approximations. More accurate analyses will necessitate more accurate quantification of model components and, where applicable, mined-site specific input data and empirical models such as measurable SOC.

Conclusions

Forest establishment on reclaimed mined land could be used as an efficient means to sequester significant amounts of atmospheric CO₂ in the form of vegetation biomass, SOC, litter layer, wood products in use and in landfills. A mined land of average site quality (SI = 25 m) that is planted to mixed hardwood forest that is tilled and fertilized, and treated for weed control would sequester about 235 Mg C ha⁻¹ at the end of the first 60-year rotation, while the net carbon stock measured at the end of the second 60-year rotation (120 years following forest establishment) would be approximately 168 Mg C ha⁻¹. A compelling argument could be made that coal mining companies should restore the forests that existed on the land prior to mining, as opposed to creating grasslands that might be abandoned in the future. The greater good of proper application of mined land reclamation and reforestation practices would be that previously existing healthy forest ecosystems will be restored, which would help sequester the carbon that was once released in the atmosphere during coal mining and fossil fuel burning for energy.

The main outcome from this work was the first approximation of a decision support system based on expert knowledge for predicting carbon sequestration for different forest establishment practices on mined land. The DSS could be used by mined land owners to choose the most appropriate forest establishment and management practices that would maximize carbon sequestration on mined land. There are a number of interactive selection options in the DSS that could be adjusted to match the mined land area, site quality, local timber market (to plan for pulp or sawtimber production), to mention a few, that would correspond to the mined land conditions and timber production goals for most mined land owners in the coalfields of the eastern United States.

As additional data become available, we intend to constantly improve the performance of the proposed DSS by incorporating the latest findings from the literature regarding the transformation and transfer of terrestrial carbon through each phase of the carbon cycle on mined land, from vegetation biomass accumulation to carbon sequestration in the soil and wood products in use and in landfills. In the future, as we develop better and improved iterations of the DSS, it could be used by mined land planners to recommend appropriate reforestation and forest management scenarios to mining companies and mined land owners throughout the contiguous United States.

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TASK 2: Develop classification and inventory criteria and procedures for mined land.

Executive Summary

Successful establishment of a productive forest on reclaimed mined land can provide economic benefits through wood production, wildlife habitat, watershed protection, and carbon sequestration. The SMCRA of 1977 requires that reclaimed land be equally as, or more productive than pre-mined conditions. However, since the passage of this law, few productive forests have been established due to poor mine soil conditions, lack of incentives for mine operators to plant trees, and inability to estimate mine soil quality for forests.

We developed a mine soil productivity model that incorporated key soil properties most influenced forest growth on mine soils:

$$PI = (pH \times \text{texture} \times \text{density} \times CF)^{1/4} \times \text{depth} \quad \text{Eq. 1}$$

where PI = Productivity Index; pH = sufficiency of pH; texture = sufficiency of texture; density = sufficiency of soil density; CF = sufficiency of coarse fragments; and depth = sufficiency of rooting depth. A PI was calculated for each of 52 white pine sites located in VA, WV and KY using the measured values for pH, texture, density, CF, and depth. PI values were regressed with white pine SI to determine the extent to which the general PI model correlated with SI. The fit of the general PI model to SI of the validation sites resulted in an R^2 value of 0.63. This shows that the general soil-based PI model could be used in lieu of SI to estimate the productivity of mined land for white pine.

In order to assess the influence of mine soil properties on different hardwood species, a selected site in Buchanan County, VA, was used to test a hardwood-based model. Ten mine soil properties were correlated with height growth of planted hardwood species in their third growing season. Multiple linear regressions were used to select the most influential independent variables.

Chestnut oak (*Quercus prinus* L.), red oak (*Quercus rubra* L.), and white oak (*Quercus alba* L.) are all desirable species for mast production and saw-timber production on reclaimed mines in the Appalachian region. A loam texture appears to be better than sandy loam for height growth of these species, confirming that the increased water retention of the heavier soils is likely beneficial for hardwood production. A higher pH is also correlated with red oak height growth and represents the need for better nutrient availability with this species as compared to others. Density was the only variable that correlated with white oak height growth.

Density was the common variable found to be influential in the height and diameter growth of all other species (red maple, sugar maple, white ash, sycamore). This confirms that increased soil density continues to be the major limitation and most influential soil property on tree productivity on mined lands in the Appalachian region.

An approximate PI model for hardwood productivity in the Appalachian region is:

$$PI_{HW} = (\text{density} \times \text{texture} \times RF \times \text{aspect})^{1/4} \times \text{depth} \quad \text{Eq. 2}$$

where PI_{HW} = the hardwood seedling Productivity Index, the geometric mean is taken of the product of the sufficiency of density class or D_b , texture, RF volume %, and aspect, and then multiplied by the sufficiency of depth.

These models can be used by foresters and reclamation managers to classify mine soils into a set of forest site quality classes (FSQC) for predicting site index by tree species. Soil texture,

density, and rooting depth were the most influential properties for white pine growth on post-SMCRA reclaimed surface mines, with soil density being the most important. The models can be used for mapping mined landscapes and making management decisions. Ordination symbols can be used to recognize the most limiting properties and offer suggestions of species selection and silvicultural prescriptions for land managers. Observations of vegetation type and vigor will lend much insight into determining map unit boundaries.

Our FSQC models should aid mine operators, foresters, and landowners in determining the productive capability of mined land, in making management decisions, and in reducing the risk associated with planting trees on mined land.

Experimental

General Productivity Index Model Development and Validation

Based on previous research, nine soil and site properties were selected for inclusion in a general forest PI model. Sufficiency curves defining the relationship between tree or root growth and levels of the soil and site properties were developed based on past and current research. The general PI model was developed based on the mathematical format of Equation 2, except that only one soil layer (0-20 cm) was analyzed, different properties were measured, and site properties were included together with soil layer properties.

Nine soil and site properties were measured or estimated to validate the general PI model for white pine growth on post-SMCRA reclaimed mine lands. Fifty-two white pine stands ranging from 10 to 18 years old were sampled at sites in Wise County, VA, and Nicholas, Mercer, and Wyoming Counties, WV, for soil and site variables and for site index (SI = dominant tree height at age 50) of white pine, as explained below.

Field-Measured Soil and Site Properties

The pH, EC, texture, color, sandstone %, RF volume percent, soil density class, potential rooting depth, slope, and aspect were estimated at several (3 to 5) locations in a sample area and averaged for a representative result due to the extreme heterogeneity of mine soils. The color data was not included in the model but was used as ancillary data. The sample area varied from 9 to 36 m² depending on tree and stand diversity and the uniformity of soil types.

The dominant soil material in the upper 20 cm was evaluated at each sample site. The pH and EC were measured with a Hanna HI 9812 field meter (Hanna Instruments Inc., Woonsocket, RI) in pH units and $\mu\text{S cm}^{-1}$, respectively. The soil hue, value, and chroma were recorded using Munsell Color Charts. On sites where an A horizon had formed on the surface, the Munsell color was read below the zone of apparent organic matter (OM) accumulation. Soil texture class was estimated by rubbing moistened soil. Sandstone % was estimated as the proportion of sandstone fragments compared to the total volume of all rock fragments (the RF volume) in the sample area.

No completely quantitative method was found to accurately predict mine soil D_b because of the high volume of RFs. Conventional D_b measurements require laboratory calculations of RF volume and moisture content, and are too time-consuming for field practical measurements. Therefore, the “density class” of the upper 20 cm was estimated based on the average penetration depth of a sharpshooter (tapered shovel with rounded tip blade 14 cm wide and 40 cm long) along with observations of soil rupture resistance and RF type and volume. The sharpshooter

was stepped on using a steady force from the weight of a 70 kg person, and the depth and ease in which it penetrated the soil was noted along with the associated soil properties listed above. The following guides were used to estimate five general density classes: if the sharpshooter penetrated easily to 25 cm or more, then a density class of “very low” was assigned; if penetration was 16 to 25 cm with slight resistance, then a density class of “low” was assigned; if penetration was less than 15 cm with moderate resistance, then a density class of “moderate” was assigned; if penetration was less than 5 cm with strong resistance, then a density class of “high” was assigned; and if penetration was less than 2 cm then a density class of “very high” was assigned. The density class was decreased one class in soils with an estimated RF content greater than 50%, provided that the moist rupture resistance (a.k.a. moist consistence class) at the depth of maximum sharpshooter penetration was not very firm or extremely firm (Soil Survey Division Staff, 1999) as confirmed by shallow pit excavations. In moist soils with low RF content and textures finer than sandy loam, the density was increased one class because those soil conditions allow sharpshooter penetration into soil that has moist rupture resistance of very firm or extremely firm as confirmed by shallow pit excavations. In extremely dry soils, no adjustment was made. Along with the rupture resistance, fine root growth widely-spaced or matted between aggregates and large aggregate size were used to confirm that the soil was dense.

The potential rooting depth (cm) was determined by using a screw auger (round tip screw head 16-cm long and 5-cm wide with 3 complete turns and on a 97-cm long shaft) and turning it into the ground until significant resistance (more than upper body strength was required) was felt or complete refusal was reached. Layers with “bridging voids” (large air gaps between rocks), greater than 90% rock fragments, and essentially no soil were considered root limiting, along with dense, compacted layers.

Site factors were measured at each sample location. Percent slope was measured using a standard clinometer. Aspect was measured as an azimuth on slopes greater than 15% using a standard compass.

At each sample point the nearest two to four trees were measured using the growth intercept model developed by Beck (1971), in which the length of the first five internodes (distance between whorls of branches) beginning at breast height (1.4 m) is measured and converted to a site index (Equation 3). Waiting until the tree reaches breast height minimizes the effects of strong competition by ground cover on tree seedlings.

$$SI = 26 + 6.6 (5\text{-year intercept length}) \quad \text{Eq. 3}$$

where SI = white pine site index (predicted tree height in feet at age 50); 26 and 6.6 are coefficients; and 5-year intercept = total length in feet of the first five internodes beginning at breast height.

Statistical Analysis

Multiple linear regression techniques were used in SAS 9.1 (2003) to identify the soil and site properties from the soil and site variables at each sample site that were the most significantly related to white pine SI calculated in Equation 3. Transformations of the independent variables were used to linearize the data based on known relationships. Multi-collinearity assessments were made using variance inflation factors (VIF's) (Montgomery et al., 2001). Data points with large influence or leverage on the model were identified using various influence statistics (Montgomery et al., 2001). Distributions, normality, and homogeneity of variance of the data

were all analyzed using residual plots, stem-leaf plots, and normal probability plots (SAS 9.1, 2003). Mallows' C(p) statistic was used as a selection procedure to derive a list of the best models (Montgomery et al., 2001). Importance factors (IF) for each variable were calculated using the absolute value of standardized coefficients (Montgomery et al. 2001), and normalizing the values from 0 to 1. The PI model was developed and modified from Gale (1987), and regressed with SI using Microsoft Excel.

Sufficiency curves were developed for nine soil and site properties that were reported in the literature to have had significant effects on tree growth, and could be analyzed in the field. Many of these curves have previously been adapted for use on mine soils and for tree growth as opposed to agricultural crops. Sufficiency curves for pH, EC, D_b , rooting depth, and slope have been previously established for white pine productivity measurements of mine soils in the study region (Andrews 1992; Torbert et al. 1994). Sufficiency curves for RF, texture, aspect, and sandstone % were developed based on previous research.

Mapping

A mountaintop removal mine was selected as a mapping demonstration area to test the practicality of the classification system developed. It was located in Dickenson County, VA and is known as the Flint Gap site. This site was reclaimed in 1994, and herbaceous vegetation along with some planted white pine and Virginia pine (*Pinus virginiana* Mill.) dominated the site. Some volunteer seedlings had become established as well, but very little growth was evident.

Data were collected in selected locations using standard soil mapping techniques, and the model developed was used to delineate polygons that represented different FSQC. Ordination symbols of p, r, t, and c were designated to map units if one of the selected properties of pH, RF, texture, or density respectively had a sufficiency value of 0.6 or less. A rooting depth of less than 50 cm was given the ordination symbol of d. Spot symbols were used to indicate wet spots on the landscape.

Results and Discussion

General Productivity Index Model Assessment

A sufficiency curve was developed for pH (Fig. 2-1), similar to the one used by Andrews (1992), and was adjusted using research results from Gale et al. (1991) and Torbert et al. (1990). A pH between 4.5 and 5.8 was considered optimal for white pine and was assigned a sufficiency level of 1.0, while a linear decline on each side of the optimal plateau results in a pH of 3.0 and 8.0, having a sufficiency level of 0.2. High pH values (> 7) may be enough to reduce the availability of boron (B), copper (Cu), zinc (Zn), iron (Fe), and manganese (Mn) (Brady and Weil 1999). A lower pH negatively affects the growth of herbaceous ground cover seeded during reclamation, reducing the competition with trees.

A sufficiency curve for EC developed by Andrews (1992) was used in this study (Fig. 2-2). An EC value less than 0.5 dS m^{-1} was not suspected to have an affect on white pine productivity, and the curve declines linearly to an EC of 2 dS m^{-1} and a sufficiency level of 0.2. Andrews et al. (1998) found that total soluble salts ranged from 0.02 and 1.97 dS m^{-1} across 78 mined sites. When values exceeded 1.00 dS m^{-1} , total salts became one of the most important chemical properties affecting white pine growth on mine soils.

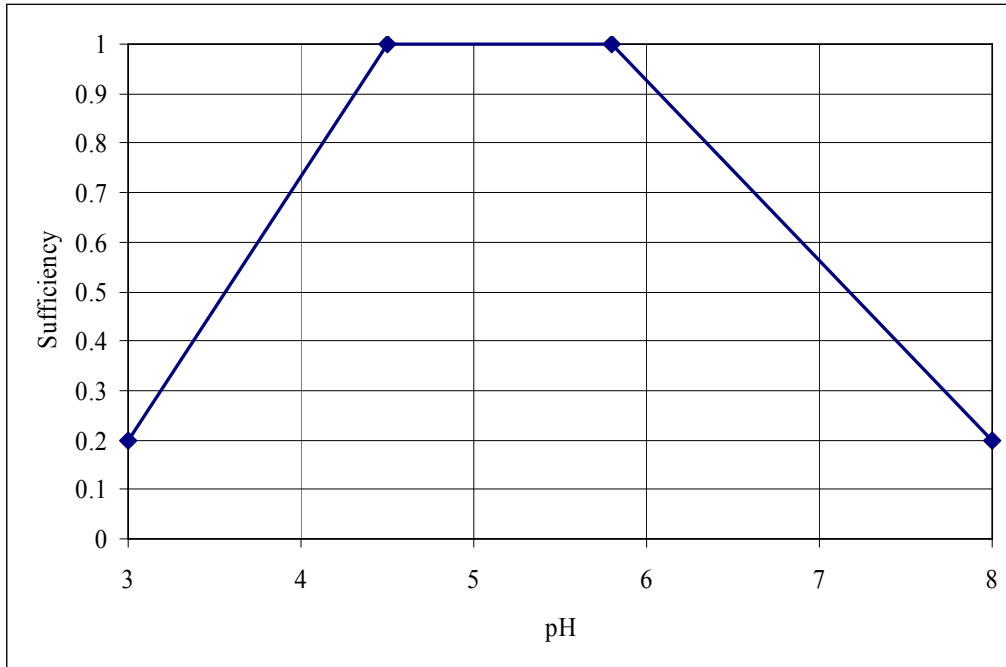


Figure 2-1. A sufficiency curve for pH was developed based on research by Andrews (1992), Gale et al. (1991), and Torbert et al. (1990).

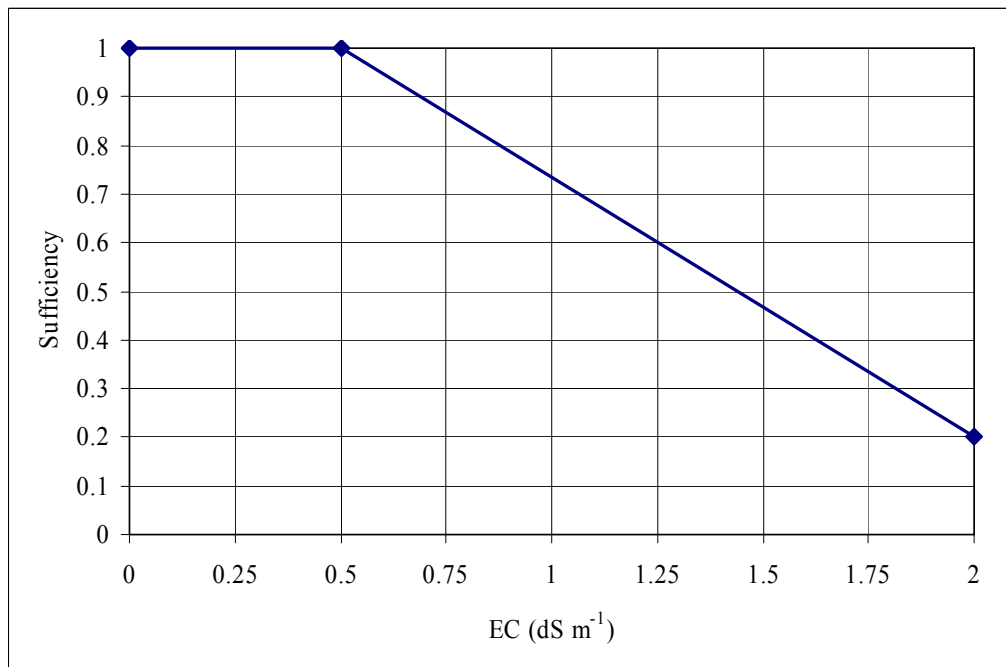


Figure 2-2. Sufficiency curve for electrical conductivity (EC) on mine soils in the Appalachian region (reproduced from Andrews, 1992).

Neill (1979) and Andrews (1992) produced sufficiency curves for D_b , both of which decline in sufficiency level above a critical D_b . The sufficiency curve developed in this study follows the same pattern but is shifted slightly to the left to correlate it with our sharpshooter

penetration method of soil density class assessment (Fig. 2-3). A point along the soil density continuum is chosen to determine a sufficiency value. Mine soils typically have less structure and porosity and fewer interconnected pores than native soils, which leads to lower soil moisture availability and aeration. The densities that were considered limiting were adjusted downward based on the D_b measurements and to account for inherently lower porosity in mine soils than in native soils. The D_b sufficiency curve was modified by Andrews (1992) using data from Pierce et al. (1983) to account for three different general particle size classes. Our method for determining soil density class in the field accounts for soil texture and allows the use of only one sufficiency curve.

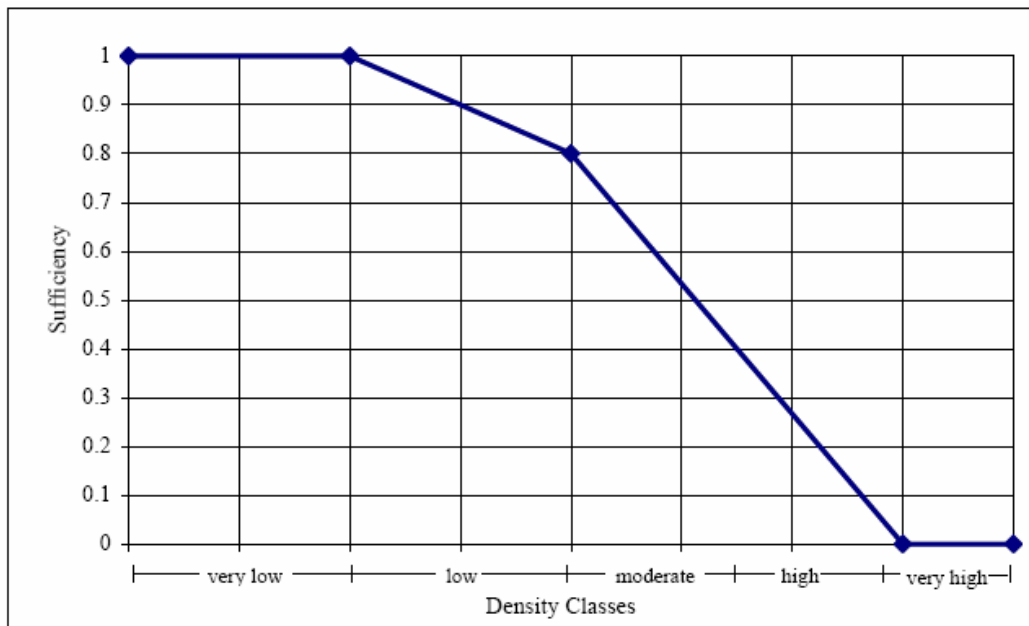


Figure 2-3. Bulk density sufficiency curve developed by Andrews (1992) and Neill (1979) and modified to accommodate the sharpshooter penetration density classes adjusted for porosity differences in mine soils compared to native soils.

With the requirement of returning the land to AOC, reclaimed mine spoil is graded with large equipment. Slopes $> 25\%$ are difficult to traverse with large equipment and the soils on steep slopes are consequently less compacted and have a deeper rooting depth than soils on flat areas (Andrews et al. 1998). Therefore, the slope of a site may be used as a surrogate for the degree of compaction. The sufficiency curve for slope developed by Andrews (1992) assigns a sufficiency of 1.0 to all slopes greater than 35% (Fig. 2-4).

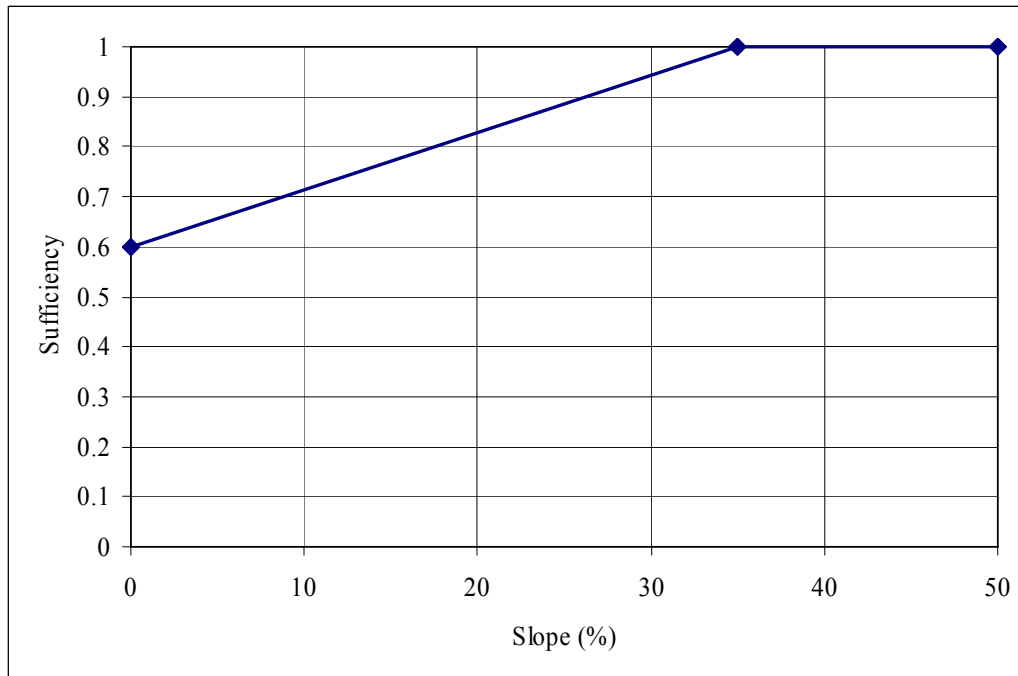


Figure 2-4. Sufficiency curve for slope on mine soils in the Appalachian region (reproduced from Andrews, 1992).

No sufficiency curves for soil texture have been published. A first approximation of a soil texture sufficiency curve (Fig. 2-5) was based on mine soil research by Burger and Zipper (2002), and on white pine growth on native soils by Lancaster and Leak (1978). High clay soils are known to be unproductive for white pines, and extremely sandy soils have low water holding capacity. Sandy loam textures are optimal for pine growth (Burger and Zipper 2002); this textural class falls within the range of silt + clay % that has a sufficiency of 1.0. Silt + clay % overlap texture class boundaries. Some growth is expected at 0 and 100 % silt + clay. Silty soils and soils with high clay content are also more easily compacted and less aerated than soils dominated by sand-sized particles. Poor aeration and drainage are chief causes of poor tree survival and growth.

The RF sufficiency curve is based on research by Rodrigue and Burger (2004). A linear relationship with increasing RF contents and decreasing sufficiency levels is expected at RF content greater than 35 % (Fig. 2-6). Bramble (1952) reported that at least 20% of soil-sized particles must be present for trees to survive. Others have recognized 90% rock fragments as being totally root limiting (John Sencindiver pers. comm. 2005).

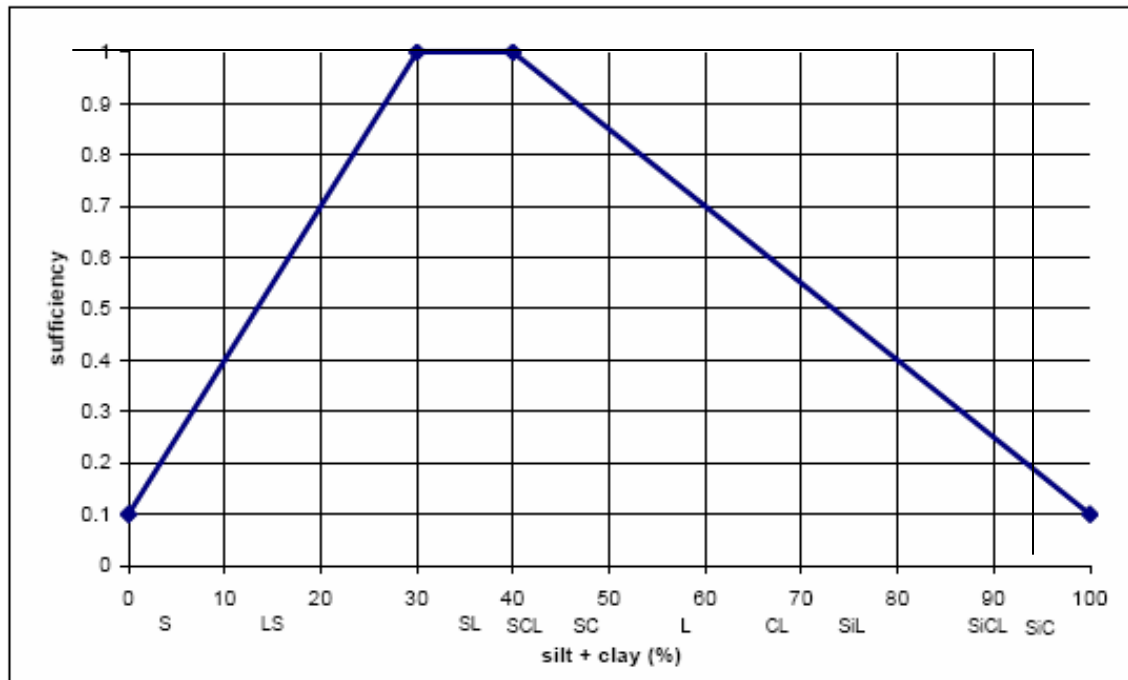


Figure 2-5. A sufficiency curve for texture and its influence on white pine growth on mine soils in the Appalachian region developed based on research from Burger and Zipper (2002) and Lancaster and Leak (1978). Silt + clay % overlap texture class boundaries.

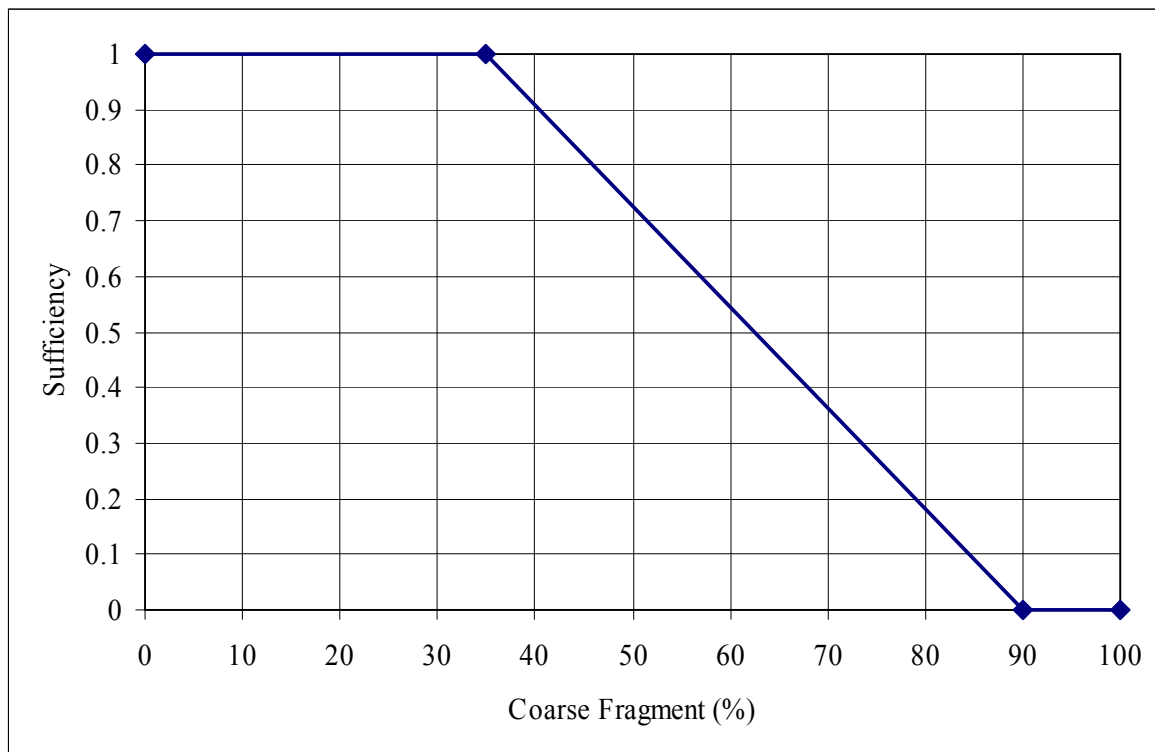


Figure 2-6. Rock fragment sufficiency as a function of rock fragment volume.

The sufficiency of potential rooting depth was defined by Equation 4 (Gale, 1987):

$$Y = 1 - \beta^d \quad \text{Eq. 4}$$

Where Y = cumulative root fraction from the soil surface to soil depth d (cm); and $\beta = 0.96$, an estimated parameter used by Torbert et al. (1994) for white pine.

The sufficiency curve for rooting depth attributes greatest importance to the thickness of the surface soil layer, with the relative importance of rooting in subsoil layers decreasing exponentially with depth (Fig. 2-7) (Gale, 1987). Torbert et al. (1988b) found that the rooting volume index (RVI = rooting depth x percent fraction < 2 mm) accounted for almost 50% of the variation in tree height for eight-year-old white pines.

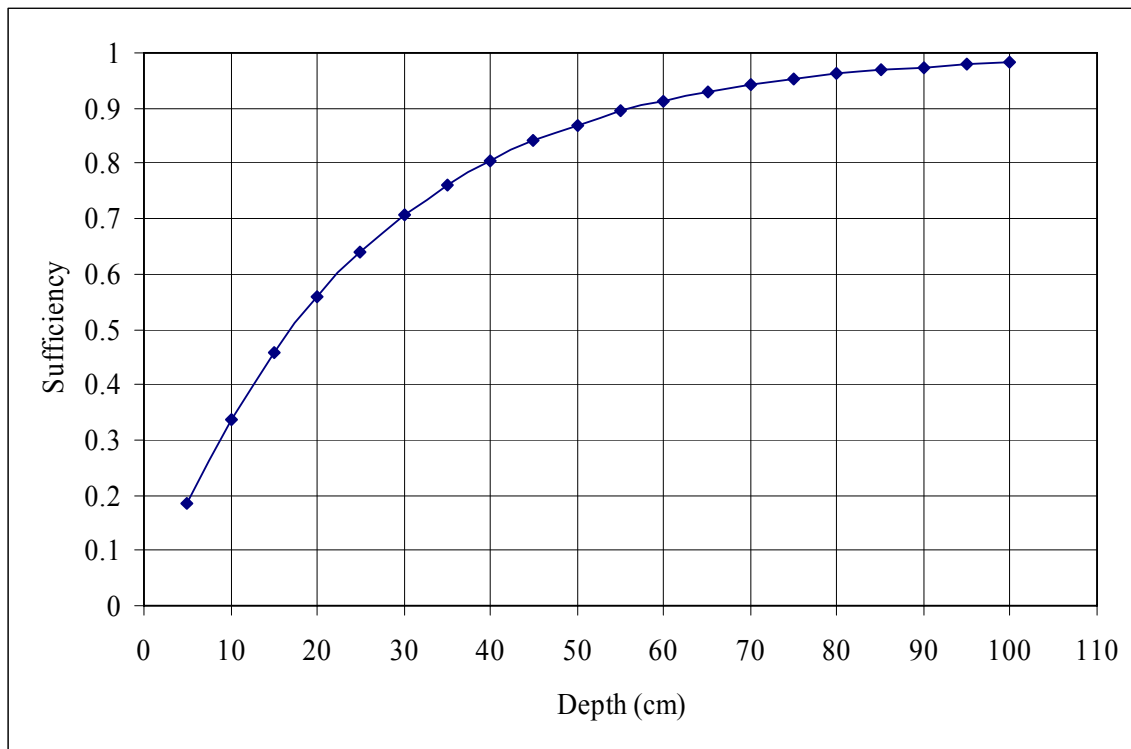


Figure 2-7. Sufficiency of rooting depth potential declines exponentially with decreasing depth (Gale, 1987).

No sufficiency curves have been previously developed for sandstone %, so a linear sufficiency curve was developed based on research from Torbert et al. (1990) (Fig. 2-8). A sufficiency of 0.4 is given for 0% sandstone because lack of sandstone is not expected to totally limit tree growth. Siltstone and shale weather into finer particles and are generally more susceptible to compaction, have fewer macro-pores, a higher pH, and higher levels of soluble salts than most sandstone spoils. In a study by Torbert et al. (1988a) of hybrid pine growth on different rock mixtures, 4-year-old trees had an average height, diameter, and volume of 146.2 cm, 40.4 mm, and 685 cm³ respectively on oxidized sandstone spoil. On siltstone spoil the values reported were 84.8 cm, 21.8 mm, and 123 cm³. After five years, Torbert et al. (1990) concluded that overall survival was not significantly affected by rock type, but tree volume was.

About 1.2 m of uncompacted sandstone material is needed to produce a mine soil of high quality and productivity for native trees (Burger and Zipper, 2002).

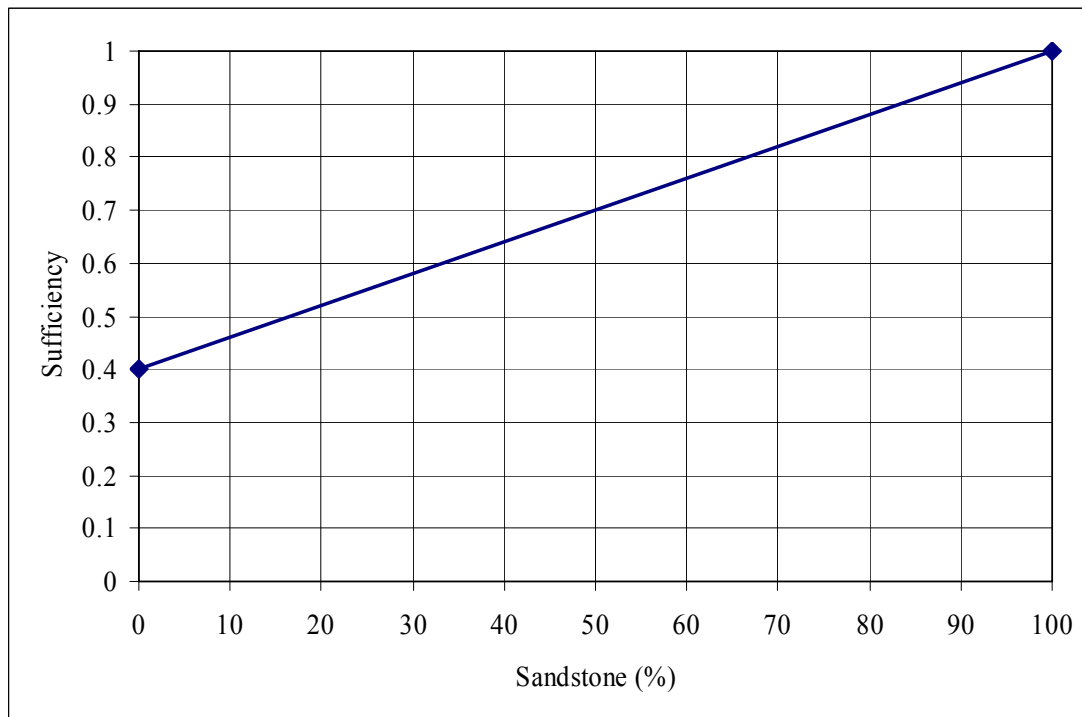


Figure 2-8. Sufficiency curve for sandstone % used on mine soils in the Appalachian region was developed based on research from Torbert et al. (1990).

A sufficiency curve was developed indicating northeast aspects being the best and southwest aspects being the worst sites for tree growth (Fig. 2-9), based on research by Hicks and Frank (1984) and Burger et al. (2002). The sufficiency levels may not be true on high-elevation mine sites, where sunlight may become limiting on steep northeast-facing aspects (Miller et al. 2004; Whittaker 1966).

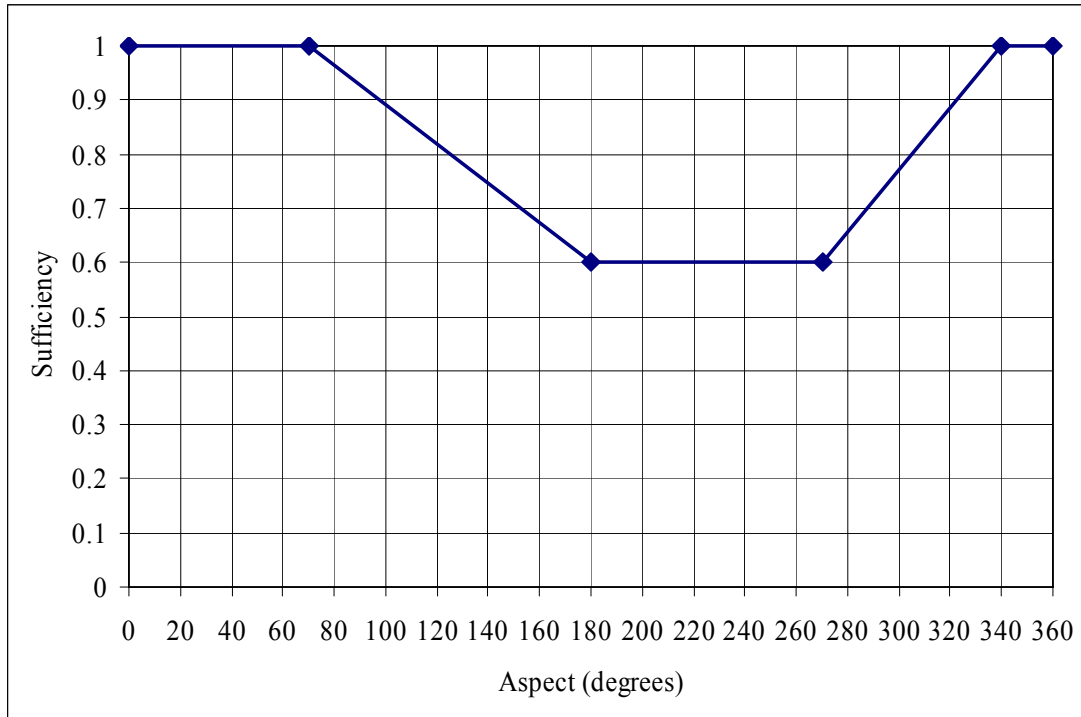


Figure 2-9. Sufficiency curve for aspect used on mine soils in the Appalachian region, based on research by Hicks and Frank, 1984 and Burger et al., 2002.

The general PI model incorporating the soil and site properties that affect forest productivity on mine soils is:

$$PI = (pH \times EC \times \text{density} \times \text{slope} \times \text{texture} \times RF \times \text{sandstone \%} \times \text{aspect})^{1/8} \times \text{depth} \quad \text{Eq. 5}$$

where PI = Productivity Index; pH = sufficiency of pH; EC = sufficiency of electrical conductivity; density = sufficiency of density class; slope = sufficiency of slope; texture = sufficiency of texture determined by silt + clay %; RF = sufficiency of rock fragment volume; sandstone % = sufficiency of sandstone %; aspect = sufficiency of aspect; depth = sufficiency of potential rooting depth (equivalent to the WF variable in Equation 2).

PI was calculated for each of the 52 white pine sites using Equation 5. PI values were regressed with white pine SI to determine the extent to which the general PI model correlated with SI. The fit of the general PI model to SI of the validation sites resulted in an R^2 value of 0.61 (Figure 2-10). This shows that the general soil-based PI model could be used in lieu of SI to estimate the productivity of white pine on post-SMCRA mined land.

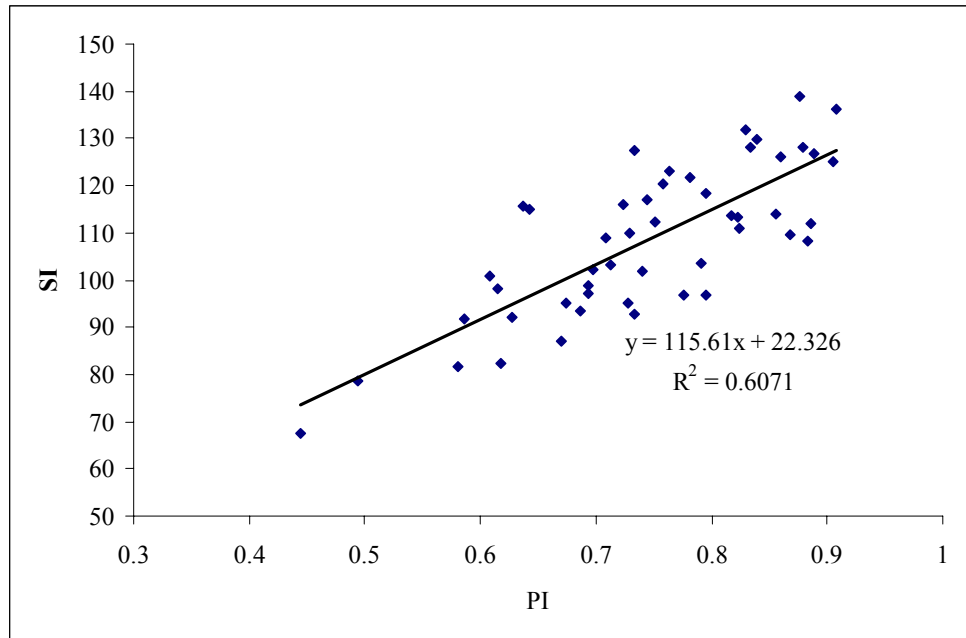


Figure 2-10. The general productivity index (PI = x) regressed with site index (SI = y, tree height at age 50) of white pine growing on mine soils in the Appalachian region.

White Pine-Specific Productivity Index Model

The general PI model (Eq. 5) calculated a geometric mean for all soil property sufficiency levels, assuming that each had the same level of importance on the PI. Soil and site properties have varying influence on productivity dependent upon the tree species. The general PI model could be improved if each property was weighted based on the extent to which it influenced growth of a particular tree species. A white pine-specific PI model was developed using regression relationships between white pine SI and soil and site properties found on the 52 measured sites.

Data from three of the original 52 sampling points were discarded because the SI values were extreme outliers, or had large influence and leverage on the model determined by influence statistics. Site index was regressed with all soil and site properties after the raw soil and site data were transformed to linearize them and reduce the variability. The pH variable was squared, an arcsine transformation was used on all data that was recorded in percent, and the RF and slope variables were log transformed.

The C(p) selection procedure indicated that the best white pine PI model included only the variables of texture, density, and rooting depth ($R^2 = 0.695$ and adjusted $R^2 = 0.675$). The VIF's indicated that no significant multi-collinearity problems existed.

Soil density was the most significant variable ($p < 0.0001$) affecting tree growth (Table 2-1), as predicted by the work of Daniels and Amos (1981) and Torbert and Burger (1990). A regression of SI and soil density class alone resulted in a R^2 of 0.53, with higher densities having lower SI values. Rooting depth was the second most influential significant variable ($p = 0.0002$), which agrees with the results reported by Andrews et al. (1998) and Torbert et al. (1988b).

Rooting depth is not expected to be as important in seedling survival and early growth when the root system is not yet fully developed. Sandy loam and loam were the only textures recorded across all of the validation sites. This may have led to a biased evaluation of the texture variable, but the variable was significant ($p = 0.0051$) (Table 2-1) and has been reported as an influential property by Burger and Zipper (2002).

Table 2-1. Standardized coefficients, importance factors, and significance values for the independent variables used in the final model (Equation 6).

Variable	Standardized Coefficient	Importance Factor	<i>p</i>-value
density	-0.54219	0.47	<0.0001
rooting depth	0.36684	0.32	0.0002
texture	-0.24362	0.21	0.0051

The pH variable was insignificant ($p > 0.10$). The soil reaction ranged from pH 4.3 to 8.0 with the distribution of values skewed to values lower than the median value (Table 2-2). Most native trees in the Appalachian Mountains grow where pH is approximately 5.5 (Skousen et al., 1994) but some species can also grow well at more neutral pH values. A more diverse range of observed pH values would have likely increased the importance of pH on the model.

Table 2-2. Ranges of measured values and sufficiency values for pH, electrical conductivity (EC), aspect, texture, rock fragment (RF) content, sandstone (SS) content, slope, soil density, and soil depth at 52 sites in southern West Virginia and southwest Virginia.

Property	pH	EC (dS m⁻¹)	Aspect (degrees)	Texture (USDA class)	RF	SS	Slope	Density	Depth (cm)
					----- (%) -----				
Range of values	4.3 - 8.0	0.01 - 0.25	1 - 355	SL, L, SiL	10 - 43	10 - 90	1 - 50	very low-high	28 - 100
Range of sufficiencies	0.2 - 1.0	1.0 - 1.0	0.6 - 1.0	0.55 - 1.0	0.86 - 1.0	0.45 - 0.94	0.6 - 1.0	0.2 - 1.0	0.68 - 0.98

RF volume % ranged from 10 to 43% (Table 2-2), which was lower than earlier reported, possibly due to the increased age and weathering time of the white pine validation sites. RF volume % was negatively correlated with SI and was an insignificant variable ($p > 0.10$). Rodrigue and Burger (2004) found RF volume % to be negatively correlated with SI of white oak, and the same was expected in this study. However, the low levels of RF volume % in this study may not be in the range in which limitations to growth occur, but they do affect water holding capacity and total rooting volume, both of which are extremely important to forest productivity (Aydelott 1978). RF volume % may be more important on younger sites for seedling survival when trees have not yet developed an extensive root system, available soil moisture is limiting, and most RFs have not undergone physical weathering into finer soil material (Haering et al. 1993).

EC was not significantly ($p > 0.10$) correlated with white pine in the white pine PI model, contrary to the results of Andrews et al. (1998) and Rodrigue and Burger (2004). In a study on 10-year-old white pines by Torbert et al. (1988b), the highest EC level recorded was 1.7 dS m^{-1} and it corresponded to a tree size of only 1.18 m. This suggests that a critical value of 1 dS m^{-1} is associated with white pine productivity and all EC values in this study were lower than 1 dS m^{-1} (Table 2-2). All textures were sandy loam, loam, and silt loam (Table 2-2), which have been reported to have low EC values, while finer textures are more commonly associated with higher EC levels (Rodrigue and Burger 2004). The ages of the sites were all between 10 and 18 years, allowing any initially high salt levels to leach over time. However, the use of the EC variable for younger sites (< 5 years) may be beneficial for predicting tree survival.

In this study slope was insignificant ($p > 0.10$), but it could serve as an indicator of probable soil density, as flatter slopes tend to be more compacted on post-SMCRA mine lands (Andrews et al. 1998). Aspect was also insignificant ($p > 0.10$) in this model. Aspect becomes more important as slope angle increases and steep, southwest-facing slopes should be the driest and thus have the lowest SI values for white pine. However, soil density decreases as slope angle increases, and therefore the lack of compaction and increased rooting depth may offset the effect of aspect on steep, post-SMCRA reclaimed mine soil slopes. The proportion of sandstone was not significant ($p > 0.10$) in this PI model. However, the proportion of different rock types in the topsoil substitute affect SI to some degree because they control the mine soil color, texture, and pH properties that occur after years of exposure and weathering.

A relative IF was calculated for each soil property in the regression model. IFs were calculated by normalizing the standardized coefficients from 0 to 1. Density was the most important soil property that affected white pine growth in this data set, followed by rooting depth, and soil texture (Table 2-1). The sufficiency level of each soil property was weighted by its relative importance (IF) as shown in the following additive PI model:

$$PI_{wp} = (\text{texture} * IF_{\text{txt}}) + (\text{density} * IF_{D_b}) + (\text{depth} * IF_d) \quad \text{Eq. 6}$$

where PI_{wp} = white pine-specific Productivity Index; texture = sufficiency of texture; density = sufficiency of soil density class or D_b ; depth = sufficiency of rooting depth; and IF = importance factor for each soil property (Table 2-1).

A regression of PI_{wp} with SI (Fig. 2-11) shows that weighting the sufficiency values based on the relative importance of each soil property improved the mine soil productivity estimation. The R^2 of the PI_{wp} versus SI relationship was 0.68, better than the R^2 of 0.61 for the general PI model (Fig. 2-10).

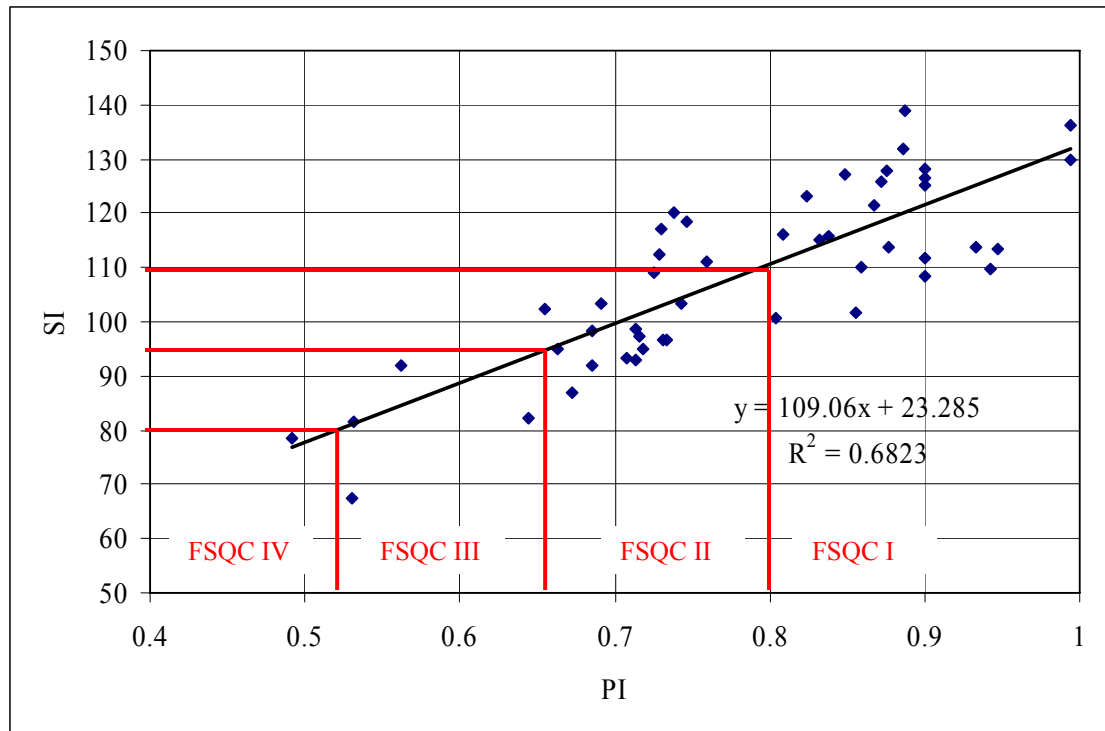


Figure 2-11. A regression of the white pine-specific productivity index ($PI_{wp} = x$) with site index ($SI = y$, tree height at age 50) of white pine (*Pinus strobus* L.).

Forest Site Quality Class Development

For management purposes, foresters commonly divide the site quality gradient found across the landscape into site quality classes. The PI_{wp} was used to separate five categories of FSQC for white pine, with FSQC (I) being the most productive and FSQC (V) being the least productive (Table 2-3). No white pines were found in this study that survived in soil-site conditions of FSQC (V). The SI breakpoints for white pine were based on the research of Doolittle (1958), who found the average SI for white pine on natural soils in the southern Appalachians to be 24 m. His study showed an SI range from 20-30 m.

Table 2-3. Productivity index (PI) is associated with forest site quality classes (FSQC) and predicted site index (SI, tree height at age 50) for white pine growing on mine soils in the Appalachian coalfield region.

PI	FSQC	SI (ft)	SI (m)
≤0.38	V	< 65	< 20
0.39 - 0.52	IV	65-79	20-24
0.53 - 0.66	III	80-94	24.4-28.7
0.67 - 0.80	II	95-110	29-33.5
>0.80	I	> 110	>33.5

The following example data can be used to demonstrate the use of the FSQC to predict white pine SI: silt + clay % = 60 %, density level = midrange moderate, and rooting depth = 57 cm. $SI_{wp} = (0.7 \times 0.21) + (0.5 \times 0.47) + (0.9 \times 0.32) = 0.67$. According to Table 2-3, this value falls on the high end of the range for FSQC III, and white pines growing on this site will likely have a SI in the high end of the 80 – 94 ft range.

Hardwood Productivity Index Model Development

The PI_{wp} model appears to be a good estimator of FSQC for white pine on older surface mines. However, some reclamationists may want to plant trees immediately following final reclamation grading or before bond release. We believe that the addition of the RF volume %, EC, sandstone %, and color variables would be beneficial for sites less than five years old. Sites older than this have already been through the initial weathering stages during which salts are leached and easily weathered rocks have broken down into soil fines, and the PI model similar to that discussed above may be more appropriate.

Native hardwood tree species may be preferred on some reclaimed mined sites. Hardwood species may respond differently to mine soil properties compared to white pine (Burns and Honkala 1990). Therefore, it would be important to calibrate sufficiency curves for hardwoods, to the extent possible, based on published species/mine soil relationships. Hardwoods have only recently been used for post-SMCRA reforestation in the Appalachian region, and very few sites exist for model validation. However, based on the success of this initial FSQC model developed for white pine, it appears that an adequate general model could be developed for hardwoods as well. Furthermore, hardwood SI can be estimated with site index comparison curves developed for several Appalachian species (Doolittle 1958).

Hypothesized sufficiency curves have been developed for hardwood species based on known silvicultural characteristics of species native to the Appalachian region (Burns and Honkala 1990). The EC, density class, slope, and rooting depth sufficiency curves developed for white pine are considered adequate for hardwoods. The pH curve is shifted toward higher pH values since most hardwoods are not as acid-loving as white pine and other conifers (Figure 2-12). The texture curve shifts toward higher silt + clay % and has a wider optimal range (Figure 2-13). Hardwoods are not adversely affected by heavy clay soils (Lancaster and Leak 1978), but structureless mine soils will likely continue to present aeration problems when sand percent is very low. The RF curve indicates that RF volume % will become a limiting factor for hardwood

productivity at lower levels than for white pine (Figure 2-14). White pine is more tolerable of stony, droughty soils and can be productive on sites where moisture limits optimal hardwood growth (Lancaster and Leak 1978). Sandstone rock types are more acidic, weather into sandy loam soil textures, and have higher nutrient levels than siltstone rock types (Torbert et al. 1990; Burger and Zipper 2002; Haering et al. 1993). Due to a higher acceptable pH range and increased tolerance of fine textured soils, the hardwood sufficiency curve for sandstone % indicates that optimal sandstone percents may be lower than for white pine (Figure 2-15). The hardwood sufficiency curve for aspect designates a lower sufficiency rating for southwest aspect in order to capture differences in drought tolerance from white pine (Figure 2-16).

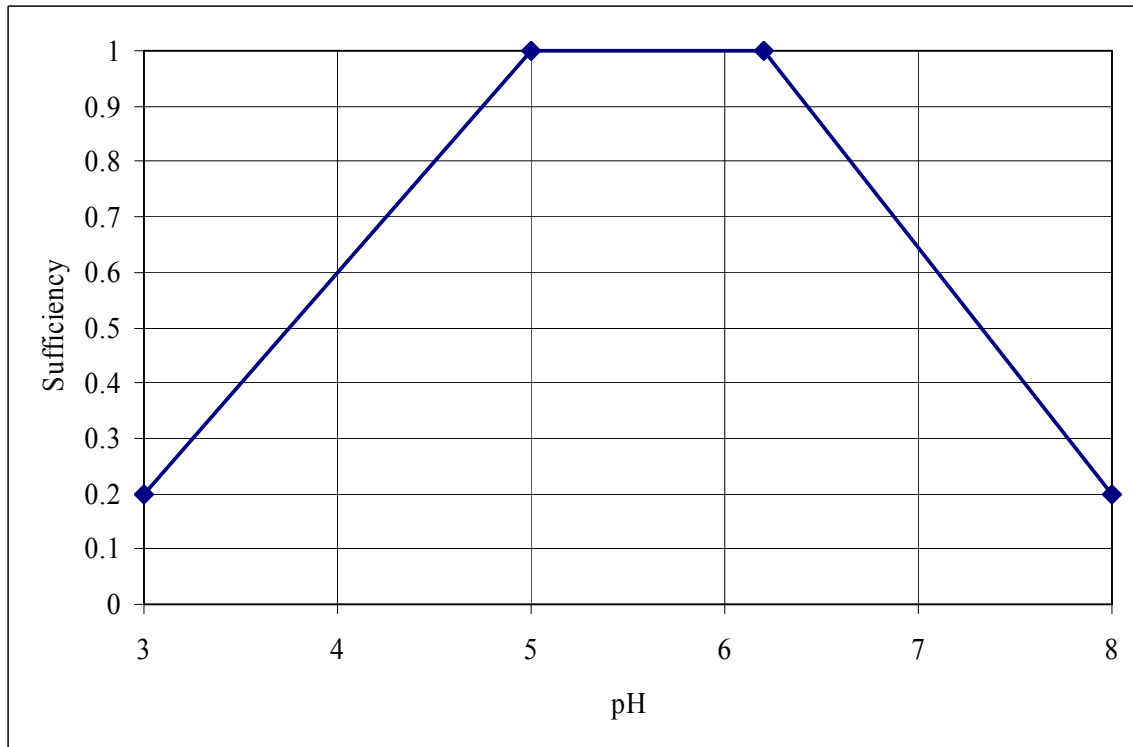


Figure 2-12. Sufficiency curve for pH used for hardwoods on mine soils in the Appalachian region.

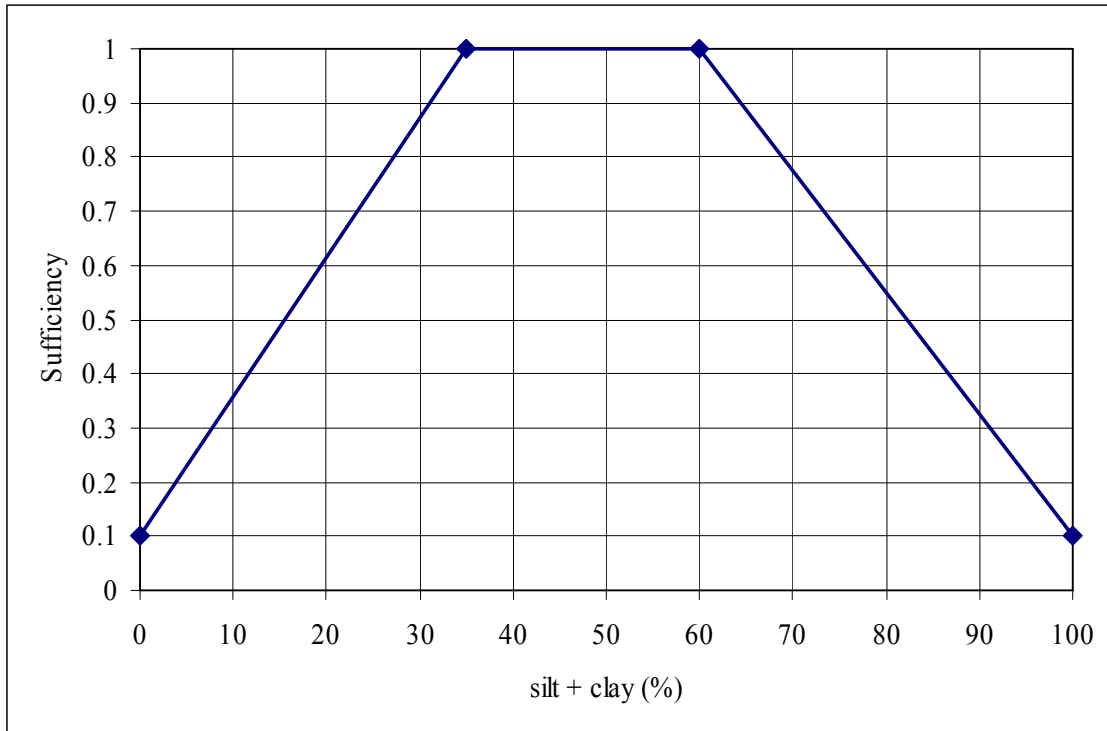


Figure 2-13. Sufficiency curve for texture used for hardwoods on mine soils in the Appalachian region.

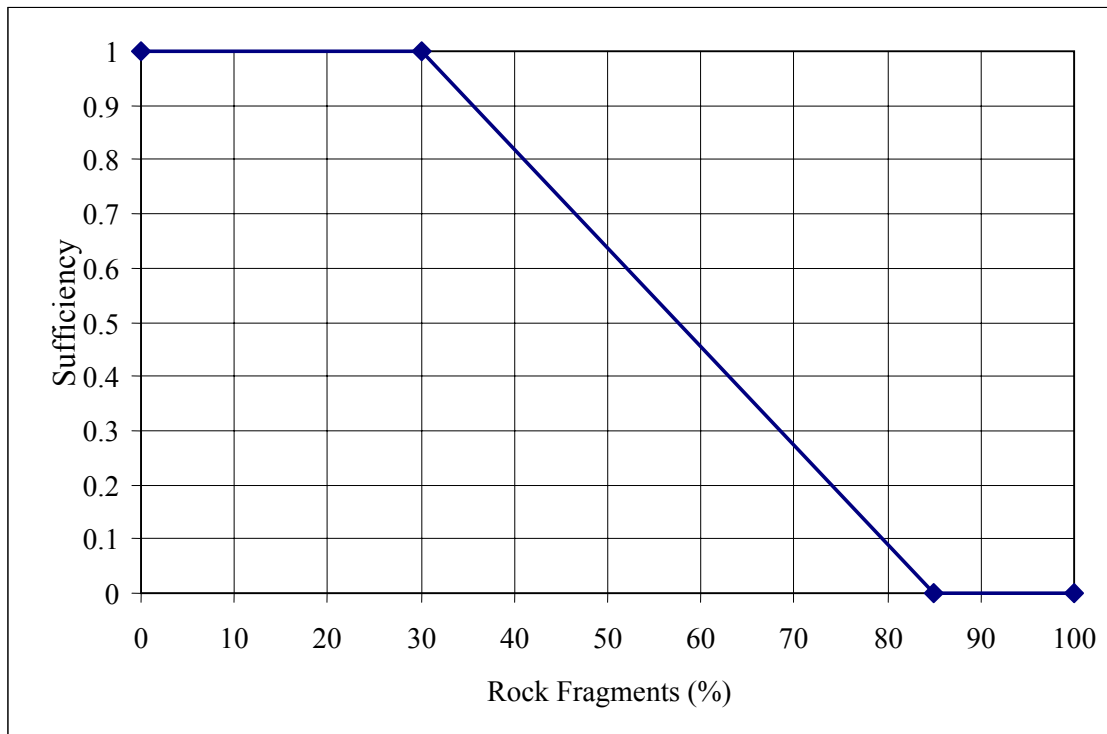


Figure 2-14. Sufficiency curve for rock fragments used for hardwoods on mine soils in the Appalachian region.

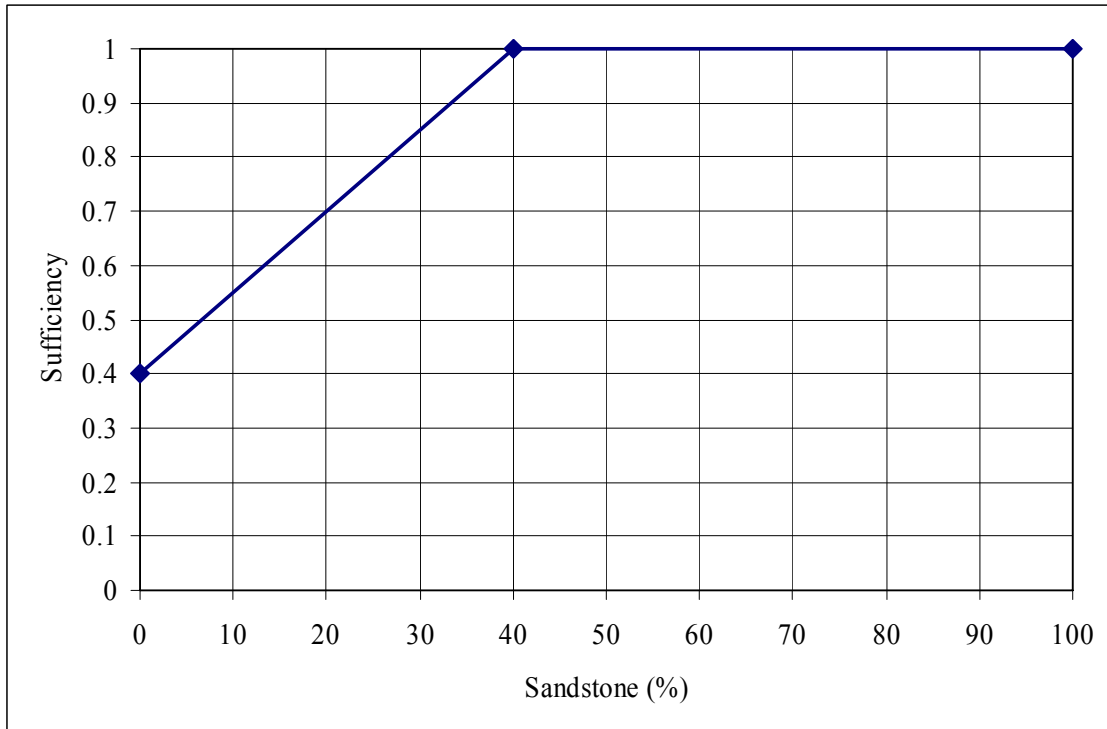


Figure 2-15. Sufficiency curve for sandstone % used for hardwoods on mine soils in the Appalachian region.

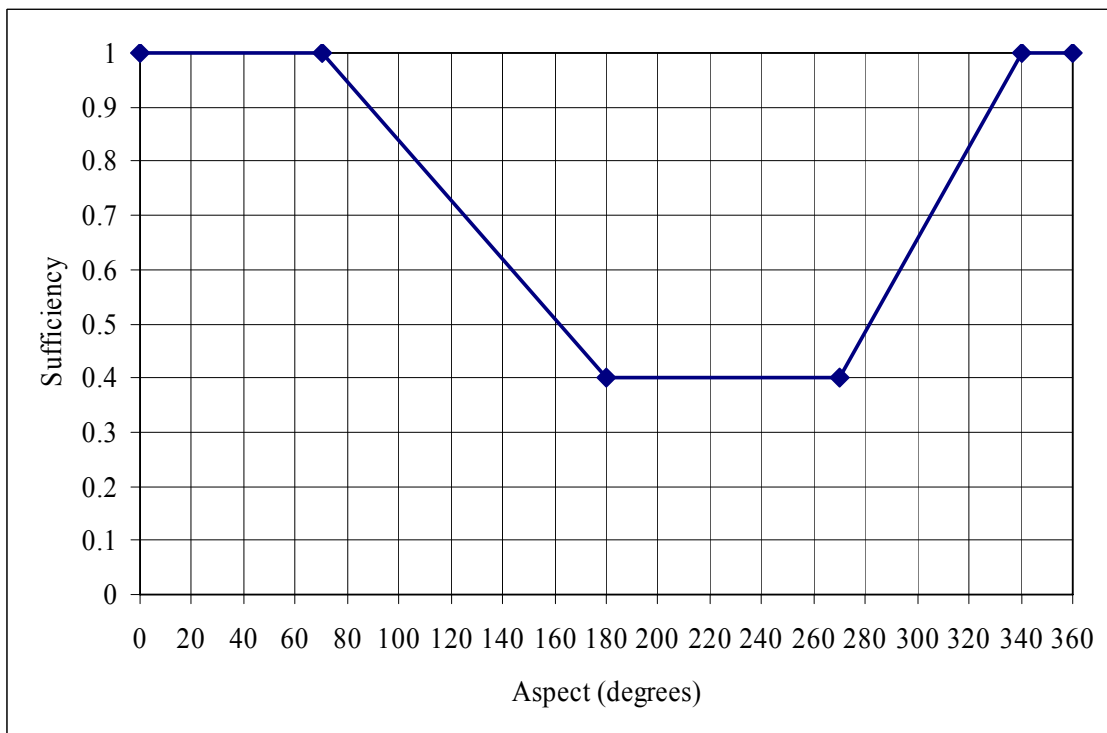


Figure 2-16. Sufficiency curve for aspect used for hardwoods on mine soils in the Appalachian region.

Rapoca Study Site

In order to assess the influence of mine soil properties on different hardwood species, a selected site in Buchanan County, VA known as “Rapoca” was evaluated for the original nine soil and site properties used in the general PI model and the soil color variable. The 10 properties were correlated with height growth of planted hardwood species in their third growing season (Table 2-4). Multiple linear regressions were performed using the properties with the highest correlation coefficients and used to select the most influential independent variables, and recognize any multi-collinearity problems. All results are interpreted with caution due to the juvenility of the tree seedlings, competition from herbaceous ground covers during the first two growing seasons, and the differences that may occur in later growth.

Table 2-4. Correlation coefficients for height growth of selected hardwood species correlated with pH; electrical conductivity (EC); rock fragments (RF); color; sandstone (SS); density; slope; aspect; and rooting depth.

Species	pH	EC	Texture	RF	Color	SS	Density	Slope	Aspect	Rooting Depth
Red Oak	0.40655 0.2775 9	-0.28236 0.4616 9	0.42348 0.256 9	0.53704 0.136 9	-0.13406 0.731 9	-0.1112 0.7758 9	-0.54789 0.1267 9	0.41411 0.2678 9	-0.3186 0.6013 5	-0.62112† 0.0742‡ 9§
White Oak	-0.00739 0.9849 9	-0.25754 0.5035 9	0.19592 0.6134 9	0.32633 0.3914 9	-0.30309 0.4279 9	0.25952 0.5001 9	-0.52352 0.148 9	0.24346 0.5279 9	-0.51659 0.3728 5	-0.50097 0.1695 9
Chestnut Oak	0.24977 0.5169 9	0.10518 0.7877 9	0.40077 0.2851 9	-0.16447 0.6724 9	0.14514 0.7095 9	-0.29733 0.4372 9	0.1139 0.7705 9	-0.12344 0.7517 9	0.2861 0.6408 5	0.27942 0.4665 9
Sugar Maple	0.27775 0.5054 8	-0.34992 0.3955 8	-0.38906 0.3408 8	0.22541 0.5915 8	0.38705 0.3435 8	0.40901 0.3143 8	-0.48467 0.2235 8	0.60187 0.1144 8	-0.78635 0.1147 5	0.04976 0.9069 8
Red Maple	0.19192 0.6489 8	-0.22954 0.5845 8	0.15438 0.7151 8	0.49422 0.2132 8	-0.36728 0.3708 8	-0.32169 0.4372 8	-0.84423 0.0084 8	0.763 0.0276 8	-0.5919 0.293 5	0.23102 0.582 8
White Ash	0.45624 0.2171 9	-0.27841 0.4682 9	-0.08335 0.8312 9	0.14753 0.7049 9	0.52744 0.1445 9	0.14847 0.703 9	-0.32749 0.3896 9	0.48932 0.1813 9	-0.16258 0.7939 5	-0.21862 0.572 9
Sycamore	0.00166 0.9966 9	-0.25819 0.5024 9	0.01858 0.9622 9	0.68024 0.0438 9	-0.13859 0.7221 9	0.1898 0.6248 9	-0.84826 0.0039 9	0.78623 0.012 9	-0.72763 0.1635 5	-0.22918 0.5531 9

† Correlation coefficient.

‡ Probability >|r|.

§ Number of observations.

Chestnut oak (*Quercus prinus* L.), red oak (*Quercus rubra* L.), and white oak (*Quercus alba* L.) are all desirable species for mast production and saw-timber production on reclaimed mines in the Appalachian region. A loam texture appears to be better than sandy loam for height growth of these species, confirming that the increased water retention of the heavier soils is likely beneficial for hardwood production. A higher pH is also correlated with red oak height growth and represents the need for better nutrient availability with this species as compared to others. Density was the only variable with any correlation to white oak height growth.

Density was the common variable found to be influential in the height and diameter growth of all other species (red maple, sugar maple, white ash, sycamore). This confirms that increased soil density continues to be the major limitation and most influential soil property on tree productivity on mined lands in the Appalachian region.

An approximate PI model for hardwood productivity in the Appalachian region follows the same form as Eq. 5. A more specific model is not able to be produced due to the lack of data, but a first approximation is found in Eq. 7:

$$PI_{HW} = (\text{density} \times \text{texture} \times \text{RF} \times \text{aspect})^{1/4} \times \text{depth} \quad (7)$$

Where PI_{HW} = the hardwood seedling Productivity Index, the geometric mean is taken of the product of the sufficiency of density class or D_b , texture, RF volume %, and aspect, and then multiplied by the sufficiency of depth. Soil density is expected to be the most important factor. With further research and location of more mature stands of hardwoods to measure, IFs should improve the hardwood model and develop it into a PI specific to a particular hardwood species.

Mapping Project Demonstration Area

The mapping of the Flint Gap site proved the usability of the developed classification system. The system appears to adequately delineate map units based on forest productivity potential, and improves on inaccuracies associated with mine soil mapping using the current USDA soil series. Approximately 78 acres were mapped with a majority of the map units being a FSQC II or III, and rooting depth along with soil density were the most commonly recognized limitations (Table 2-5). This site was a post-SMCRA mountain-top removal site, was not reclaimed to original contours, and was predominantly flat. Soil density levels were low to moderate with only very few areas having very low or high levels. RF volume % and pH were highly variable and the texture was most often loam or sandy loam.

Table 2-5. Sample point data from the Flint Gap mountain top removal site in Dickenson County, Virginia. pH; electrical conductivity (EC); aspect; slope; texture; color; rock fragments (RF); sandstone (SS); density; and rooting depth were recorded and selected properties were used to calculate a white pine productivity index (PI_{wp}), and forest site quality class (FSQC) to delineate map units. Ordination symbols are used to indicate the most limiting properties.

Point #	pH	EC (dS m ⁻¹)	Aspect (deg)	Slope (%)	Texture	Color	RF (%)	SS (%)	Density	Rooting Depth (cm)	PI _{wp} †	FSQC	Symbol‡
1	5.6	0.09	flat	§	SiL	2.5Y 4/2	20	10	low	45	0.75	II	t,d
2	7.2	0.08	flat	§	SiL	2.5Y 4/2	36	10	low	46	0.75	II	p+,t,d
3	5.5	0.08	22	18	SiL	2.5Y 4/2	30	10	low	63	0.78	II	t
4	4.9	0.08	flat	§	L	10YR 4/2	25	10	moderate	>90	0.71	II	c
5	5.4	0.06	flat	8	L	10YR 4/3	20	10	low	68	0.82	I	
6	4.5	0.03	flat	§	L	10YR 4/3	20	10	low	>75	0.85	I	
7	4.6	0.01	flat	5	L	10YR 4/4	15	10	low	>75	0.85	I	
8	6.3	0.07	flat	§	L	10YR 4/2	40	10	moderate	37	0.65	III	c,d
9	6	0.03	flat	§	SL	10YR 4/4	36	50	low	33	0.81	I	d
10	6.9	0.06	flat	9	L	2.5Y 4/3	30	10	low	45	0.79	II	p+,d
11	6.7	0.03	flat	2	SL	10YR 4/4	30	22	low	60	0.87	I	
12	5.1	0.06	flat	2	L	10YR 4/2	20	10	low	40	0.79	II	d
13	4.9	0.02	flat	§	L	10YR 4/6	10	10	very low	>75	0.95	I	
14	4.9	0.02	flat	§	L	10YR 4/6	10	10	low	40	0.79	II	d
15	5	0.02	flat	3	L	10YR 5/6	10	25	low	45	0.79	II	d
16	5.5	0.02	flat	2	L	10YR 4/4	20	10	low	72	0.82	I	
17	6	0.03	flat	§	SL	10YR 4/4	34	50	moderate	50	0.73	II	c
18	5.4	0.02	flat	3	SL	10YR 5/6	25	75	moderate	32	0.67	II	c,d
19	6	0.03	flat	5	SL	10YR 4/4	34	50	moderate	52	0.73	II	c
20	6	0.03	flat	7	SL	10YR 4/3	34	50	low	90	0.91	I	
21	5.3	0.02	flat	6	L	10YR 4/4	25	10	low	68	0.82	I	
22	5.1	0.03	flat	§	L	10YR 4/3	25	15	low	52	0.82	I	
23	5.1	0.03	flat	4	L	10YR 3/2	30	50	moderate	35	0.65	III	c,d
24	5.1	0.03	flat	8	L	10YR 4/3	30	15	low	61	0.82	I	
25	4.2	0.01	flat	4	L	10YR 4/3	30	25	moderate	36	0.65	III	c,d
26	6.2	0.07	flat	2	L	10YR 4/2	30	15	moderate	40	0.65	III	c,d
27	4.6	0.02	flat	§	L	10YR 4/4	15	15	very low	80	0.95	I	
28	5.6	0.04	flat	§	SL	10YR 4/2	40	15	low	40	0.84	I	d
29	3.6	0.01	flat	4	L	10YR 5/6	15	10	low	>100	0.85	I	p-
30	4.9	0.01	flat	§	L	10YR 4/4	30	15	low	40	0.79	II	d
31	5.5	0.03	76	20	SL	10YR 4/3	30	25	low	60	0.87	I	
32	6.4	0.03	flat	§	SL	10YR 5/4	65	90	high	10	0.43	IV	r,c,d
33	6.5	0.06	flat	§	SL	10YR 4/2	40	50	moderate	45	0.70	II	c,d
34	6.5	0.04	flat	§	SL	10YR 4/2	30	25	low	>75	0.91	I	
35	4.9	0.03	flat	§	SL	10YR 4/3	15	50	low	44	0.84	I	d
36	5.9	0.04	flat	§	SL	2.5Y 4/3	30	50	moderate	41	0.70	II	c,d
37	6	0.04	flat	§	SL	2.5Y 4/3	55	85	moderate	37	0.70	II	c,d
38	4.4	0.04	210	16	SL	10YR 4/6	30	90	low	45	0.84	I	d
39	5.9	0.03	flat	§	SL	10YR 4/4	30	75	low	>100	0.91	I	
40	6.2	0.03	flat	8	SL	10YR 4/3	60	90	moderate	21	0.64	III	r,c,d

Table 2-5. (continued)

Point #	pH	EC (dS m ⁻¹)	Aspect (deg)	Slope (%)	Texture	Color	RF (%)	SS (%)	Density	Rooting Depth (cm)	PI _{wp} [†]	FSQC	Symbol [‡]
41	4.7	0.03	flat	5	SL	10YR 5/6	30	95	low	38	0.84	I	d
42	6.7	0.06	flat	8	SL	10YR 4/4	70	90	low	17	0.75	II	r,d
43	5.7	0.01	4	16	SL	10YR 5/6	45	95	low	42	0.84	I	d
44	6.1	0.03	flat	8	SL	10YR 4/3	60	95	moderate	24	0.64	III	r,c,d
45	4.4	0.04	flat	§	L	10YR 4/4	30	30	low	>100	0.85	I	
46	6.5	0.03	flat	7	SL	10YR 4/4	40	75	moderate	34	0.67	II	c,d
47	4.7	0.02	flat	§	L	10YR 5/6	25	50	low	37	0.79	II	d

[†] PI_{wp} = (texture x 0.21) + (rooting depth x 0.32) + (density x 0.47); sufficiency values used for soil properties.

[‡] Ordination symbol given if sufficiency of soil property was ≤0.6; c=density, d=depth, p=pH, r=rock fragments, t=texture.

§ 0-1%, nearly level.

The use of established vegetation on the site proved to be invaluable for determining map unit boundaries (Figs. 2-17 and 2-18). This follows patterns of previous research in which vegetation was found to be a good indicator of soil properties and site quality (Daubenmire 1976; Jones 1991). Absence of vegetation or scattered broadleaf weeds often indicated extreme acidity, while thick, pure stands of fescue, orchard grass, and/or sweet clover indicated high pH values. Stunted, chlorotic vegetation was visible in compacted areas. A site's natural vegetation distribution begins to become naturalized and more representative of soil properties after about five years. Before this amount of time the use of vegetation as soil indicators should be done with caution.

There were areas of ponded water scattered throughout the site (Figs. 2-17 and 2-18). The ponded water usually indicates high density, impermeable mine soil at some depth, and was not always associated with footslopes and depressional areas. Landscape position commonly used to delineate soil map unit boundaries on native soils did not work on this site and is not suspected to work on other reclaimed mined lands due to different mine reclamation strategies and drastically altered hydrology.

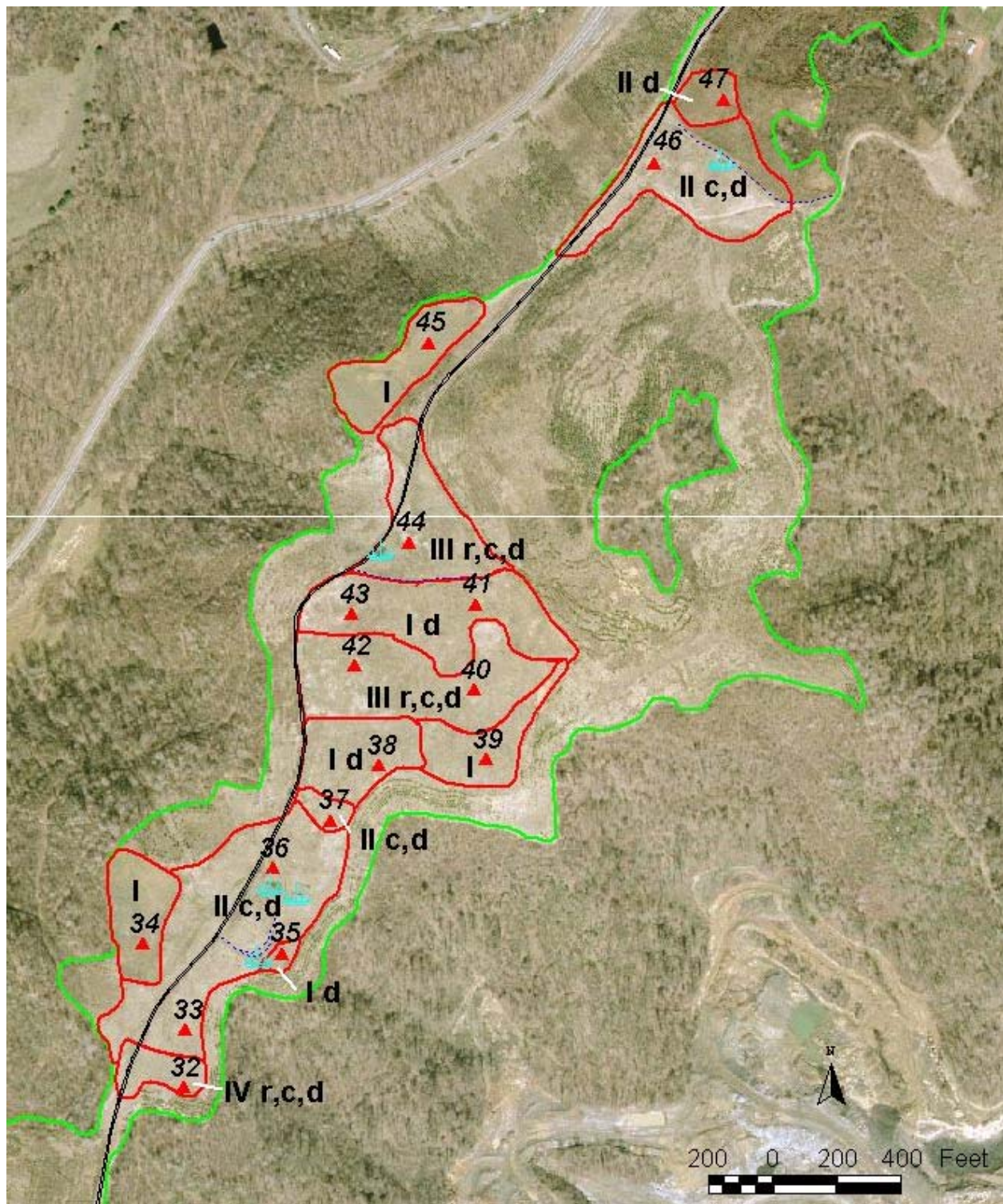


Figure 2-17. Data points taken and used along with vegetation differences to delineate map units at the north end of the Flint Gap mountaintop removal site.

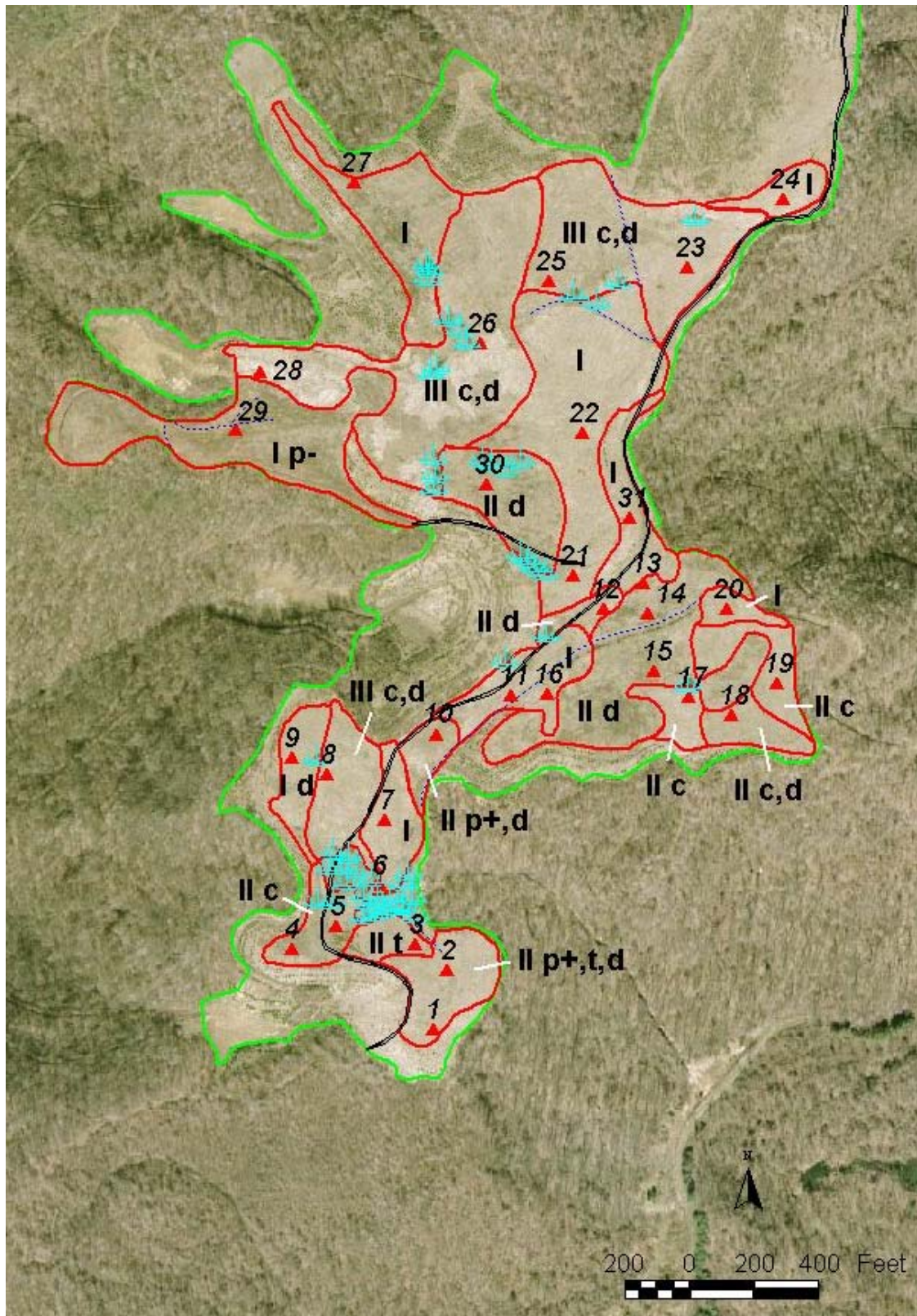


Figure 2-18. Data points taken and used along with vegetation differences to delineate map units at the south end of the Flint Gap mountaintop removal site.

The ordination symbols designated to map unit boundaries can be used for management decisions on mined land (Table 2-6). Symbols are given to a map unit if the sufficiency of the property is 0.6 or less. If the ordination symbol for density (c) is used, the land would benefit from a ripping or tillage treatment. The depth (d) symbol indicates the same, given that the ripping treatment used will reach the depth of the root limiting layer. A high RF content is represented by the ordination symbol (r), and may influence species selection decisions. More drought tolerant species should be planted in these map units. Ripping or tillage treatments may also improve these sites by bringing more soil fines to the surface and improving the planting bed. The ordination symbol for pH (p) is given a + or – to indicate which side of the optimal level the pH falls on. This will give experienced scientists an indication of nutrient availability in the soil, and may affect species selection for planting. The ordination symbol for texture (t) indicates that a soil is high in silt + clay content and will likely be limited by aeration. Different species are adapted to these sites and should be used in reclamation planting. An ordination symbol (s) (for sandstone) is suggested to be used on young sites for interpretations of rock type on a site. The ordination symbol (s) should be followed by a number 1 – 5; with 1 = < 20 % sandstone, 2 = 20 – 39 % sandstone, 3 = 40 – 59 % sandstone, 4 = 60 – 79 % sandstone, and 5 = 80 – 100 % sandstone. This will guide a land manager in making decisions on species selection and fertilization treatments based on known properties of different rock types and optimal soil properties for different tree species.

Table 2-6. Suggested management practices for ordination symbols associated with high soil density (c), shallow rooting depth (d), high rock fragment content (r), high pH (p+), low pH (p-), and high silt + clay contents (t) to optimize forest productivity.

Ordination Symbols	Management Practices
c	ripping and/or tillage
d	deep ripping
r	plant drought-tolerant species; ripping or tillage may improve planting bed by bringing more soil fines to the surface
p+	plant hardwoods or hybrid poplars
p-	plant acid-loving pines; liming may improve the site for hardwoods
t	plant FSQC II, III, and IV hardwoods; bedding may improve aeration for pines

A variety of species may be used in reclamation planting in the Appalachian region, and selecting the proper species may have dramatic consequences on reforestation success (Burns and Honkala 1990). The FSQC and mapping techniques can be used to determine which species should be planted in selected map units (Table 2-7). Map units with an FSQC of I and sometimes II suggest that white pine, red oak, and sugar maple (*Acer saccharum* Marsh.) may be the best species to plant. Tulip poplar (*Liriodendron tulipifera* L.) has been observed to have low survival rates on mine soils and should be planted only on the very best FSQC I sites. White oak, chestnut oak, and hickory (*Carya spp.*) are more tolerant of adverse soil conditions and should be used on FSQC II and some III. Green ash (*Fraxinus pennsylvanica* Marsh.), white

ash (*Fraxinus americana* L.), American sycamore (*Platanus occidentalis* L.), and red maple (*Acer rubrum* L.) are species that grow on a broad range of soil types and will likely grow better on all mined sites. These may be the only species that will grow on FSQC III and IV sites. Very little, if any, tree seedling survival and growth is expected on FSQC V and tree planting is not recommended. Any FSQC of II or higher will likely be improved with silvicultural treatments.

Table 2-7. Suggested species selection for each forest site quality class (FSQC).

-----FSQC-----				
I	II	III	IV	V
t. poplar, w. pine, r. oak, s. maple				
	w. oak, c. oak, hickory			
		r. maple, sycamore, w. ash, g. ash		
				none

Department of Energy Project: Site Quality

The FSQC for white pine was used to predict white pine productivity at each study site. These predictions will be evaluated later in the rotation and used to improve upon the FSQC. Only relationships between seedling survival percent and FSQC can be discussed at this time.

West Virginia

All three WV blocks were fairly uniform and resulted in a FSQC of III (Table 2-8). The PI values were in the lower portion of the acceptable range for FSQC III and white pine SI will likely be near 80. However, WV3 had a white pine survival percentage of nearly twice that of WV1 and WV2 (Table 2-9). WV3 had a slightly lower pH and slightly lower RF content (Table 2-8). Lower RF volume % may have improved water availability to the young seedlings and been responsible for the increased survival rates, but RF volume % are not accounted for in the FSQC model. Slightly higher sandstone percents in WV3 may have resulted in a greater portion of the total porosity being macro-pores and consequently improved aeration. Hardwood survival rates were fairly uniform across all blocks and were much higher than white pine (Table 2-9). A high density level suggests that ripping and/or tillage treatments will likely improve survival for all species. Along with high density, the high RF volume % is likely to be a limiting factor to tree growth.

Virginia

The VA blocks have more variation among blocks than do the other sites. VA1 and VA2 are identified as FSQC II, and VA3 is FSQC III (Table 2-8). White pine survival percents of VA2 are much lower than the other two VA blocks (Table 2-9), likely due to high RF volume %, low sandstone percent, and high pH. Higher pH was advantageous to herbaceous vegetation growth and the competition likely increased mortality. VA3 also had high pH but was recently reclaimed and had little established competitive vegetation. Hardwood survival followed the same pattern as white pine with VA2 being the lowest, but overall hardwood survival was greater at all blocks with VA1 and VA3 having 96% and 88% survival respectively (Table 2-9). It appears that increased competition at VA2 affected survival of all species. FSQC may prove to be inaccurate for predicting forest productivity on young sites such as VA3, because most of the measured properties will quickly change within a few years of weathering.

Table 2-8. Measured soil properties resulted in forest site quality classes (FSQC) of II and III for white pine growth at three blocks each in Nicholas County, West Virginia (WV), Lawrence County, Ohio (OH), and Wise County, Virginia (VA). pH; electrical conductivity (EC); texture; color; rock fragments (RF); sandstone (SS); density; and rooting depth were recorded and selected properties were used to calculate a white pine productivity index (PI_{wp}), and forest site quality class (FSQC). Ordination symbols are used to indicate the most limiting properties.

Block	pH	EC (dS m ⁻¹)	Texture	Color	RF (wt %)	SS (%)	Density	Rooting Depth (cm)	PI _{wp} [†]	FSQC	Symbol [‡]
WV-1	5.9	0.2	SL	10YR 4/2	55	10	high	36	0.55	III	c,d
WV-2	5.7	0.2	SL	10YR 4/2	55	10	high	32	0.54	III	c,d
WV-3	5.5	0.2	SL	10YR 4/2	45	15	high	34	0.54	III	c,d
OH-1	4.8	0.07	CL	10YR 5/4	10	20	low	45	0.76	II	t,d
OH-2	5.1	0.1	L	10YR 5/6	10	25	low	48	0.78	II	d
OH-3	6.1	0.2	L	2.5Y 5/3	10	25	low	42	0.77	II	d
VA-1	4.8	0.2	SL	10YR 5/3	30	80	low	32	0.80	II	d
VA-2	6.3	0.3	L	2.5Y 4/3	55	50	moderate	60	0.70	II	c
VA-3	6.5	0.4	SL	10YR 4/2	55	65	moderate	34	0.66	III	c,d

[†] PI_{wp} = (texture x 0.21) + (rooting depth x 0.32) + (density x 0.47); sufficiency values used for soil properties.

[‡] Ordination symbol given if sufficiency of soil property was ≤0.6; c=density, d=depth, p=pH, r=rock fragments, t=texture.

Table 2-9. First-year survival rates (%) by site (Lawrence County, Ohio (OH); Wise County, Virginia (VA); Nicholas County, West Virginia (WV)) and species type (HP = hybrid poplar; WP = white pine; HW = hardwoods).

	OH	VA	WV	Species Means
HP	49	79	32	53 B [‡]
WP	45	54	41	47 B
HW	60	81	78	73 A
site means	51 b [†]	71 a	50 b	

[†] Across rows, means that are significantly different determined by Fischer's LSD at alpha = 0.05 are followed by different lower case letters.

[‡] Within columns, means that are significantly different determined by Fischer's LSD at alpha = 0.05 are followed by different upper case letters.

Ohio

The OH blocks had finer textures than the other sites, and various topsoil depths. All three blocks resulted in FSQC II (Table 2-8), but OH3 had a much lower survival percent for white pine and hardwoods (Table 2-9). The high pH of this block (> 6) is likely responsible for tree mortality since most native species are not adapted to this range (Skousen et al. 1994). Most competitive grasses and legumes do thrive at this pH range and cause tremendous competition that likely resulted in elevated mortality levels.

White pine can compete on most soil types except for heavy clay soils (Lancaster and Leak 1978). The OH blocks had from 20% to 27% clay in the topsoil material, which are only medium levels of clay contents for natural soils in the white pine range. However, the lack of

structure in these mine soils result in reduced macro-porosity and aeration, which resembles the native soils referred to by Lancaster and Leak that have higher clay contents. Hardwoods are expected to outgrow white pine on clayey soil (Lancaster and Leak, 1978).

The compacted subsoils of all OH blocks will also affect white pine survival and growth, not only in physical root resistance, but in impeded drainage. Compacted subsoils are likely to perch water and temporarily raise the water table into the root zone. Subsurface drainage is difficult to predict in mine soils but is of extreme importance. The topography of OH1 is more undulating and will likely have better drainage than the other blocks, while OH2 is flat and has features indicating restricted surface drainage and slow percolation. White pine prefers well drained soils and cannot withstand anaerobic conditions in its root zone. Some native hardwoods are more adapted to surviving under saturated conditions and may be better adapted to these sites (Burns and Honkala, 1990). With no measure of subsurface drainage, FSQC may prove to be inappropriate for SI predictions of white pine on these OH blocks. Silvicultural operations such as deep ripping may improve drainage, and bedding can be used to raise seedlings above the local water table.

Summary and Conclusions

Many chemical and physical soil properties, as well as site factors, influence tree growth and forest productivity on mined land. Successful establishment of a productive forest on reclaimed mined land can provide economic benefits through wood production, wildlife habitat, watershed protection, and carbon sequestration. The SMCRA of 1977 requires that reclaimed land be equally as, or more productive than pre-mined conditions. However, since the passage of this law, few productive forests have been established due to poor mine soil conditions, lack of incentives for mine operators to plant trees, and inability to estimate mine soil quality for forests.

FSQC ratings based on field-measured soil properties can be used to predict potential forest productivity, which will aid in forest management prescriptions. Soil texture, density, and rooting depth were the most influential properties for white pine growth on post-SMCRA reclaimed surface mines, with soil density being the most important. Other factors may be more influential on younger sites or on sites for which native hardwoods are the intended forest type. Soil pH and rock fragment content are known to be important for forest productivity on mine soils but were not found to be significant in this study. The EC and sandstone content variables will likely be useful for recently reclaimed (< 5 years) mined sites. An evaluation of all soil properties in the general PI model is highly suggested.

Furthermore, the model developed is useable for mapping mined landscapes and making management decisions. Ordination symbols can be used to recognize the most limiting properties and offer suggestions of species selection and silvicultural prescriptions for land managers. Observations of vegetation type and vigor will lend much insight into determining map unit boundaries.

The PI model developed can only be validated where trees are present and therefore may not recognize soil properties that completely limit tree survival. Furthermore, Beck's growth intercept model may overestimate white pine SI, as extremely high SI estimations were observed in this study. Extrapolation of data beyond the ranges of soil properties, geographic regions, and PI values using this model may not be accurate.

Our FSQC model should aid mine operators, foresters, and landowners in determining the productive capability of mined land, in making management decisions, and in reducing the risk associated with planting trees on mined land.

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TASK 3: Develop reforestation methods and procedures for mined land.

Executive Summary

The Appalachian coal-producing region of the eastern United States is predominantly forested prior to surface mining. The process of surface mining removes these forests and the native soils that support them. As these lands were primarily forested prior to mining, a logical post-mining land use would be return of the land to commercial forestry uses. Three sub-studies were undertaken to determine the feasibility of establishing productive forests on mined land for products and services including carbon sequestration.

Study 1, established in 2004 in Ohio, West Virginia, and Virginia, was used to evaluate the survival and growth performance of three species assemblages under three levels of silvicultural input intended to alleviate the adverse conditions for tree survival and growth on post-SMCRA mined land. It included three forest types (white pine, hybrid poplar, mixed hardwood) and three silvicultural regimes (competition control, competition control plus tillage, competition control plus tillage plus fertilization). White pine and hardwood species grew little over the course of the first growing season, which could translate into a need for continued weed control to ensure the trees do not succumb to the competing vegetation as mean heights ranged from 25 to 40 cm for hardwoods and from 20 to 30 cm for white pine. Hardwood species survived well in WV and VA, and survived better than the other species used in OH, while white pine had the poorest survival of all species at all sites. Survival was particularly good for the site-specific hardwoods planted at each site. Weed control plus tillage may be the optimum treatment for hardwoods and white pine, as any increased growth resulting from fertilization may not offset the decreased survival that accompanied fertilization. Hybrid poplar appeared to have good potential for reverting post-SMCRA mined lands reclaimed to grasses back to forests, as this species had good growth with 50 to 65 cm of height growth in one year in WC+T at all sites and excellent growth in WC+T+F in VA. This good growth, coupled with survival percentages that may be adequate to ensure that without further weed control, an adequately stocked stand could develop, gives this species an advantage over the other species used in this study.

Study 2, established in 2001 in Wise County, Virginia, was used to determine which mine soil properties most influence white oak (*Quercus alba* L.) seedling growth, and to test whether these properties were adequately reflected in a proposed mine soil classification model developed for application in field assessments of mine soil suitability for reforestation. Seventy-two three-year-old white oaks were randomly selected across a reclaimed site in southwestern Virginia that varied greatly in spoil/site properties. Tree height was measured and soil samples adjacent to each tree were analyzed for physical, chemical, and biological properties. Our proposed mined land classification model used rock type, compaction, and slope aspect as mapping criteria. Tree height, ranging from 15.2 to 125.0 cm, was regressed against mine soil and site properties. Mapping units were not well correlated with differences in tree height. Microbial biomass, pH, exchangeable potassium, extractable inorganic nitrogen, texture, aspect, and extractable phosphorous accounted for 52% of the variability in tree growth. The regression model shows that white oaks were most successful on northeast-facing aspects, in slightly-acidic, sandy loam, fertile mine soils that are conducive to microbial activity. Nutrient availability, although found to be highly influential on tree growth, was not adequately represented in the classification model. We recommend that pH be included as a classification criterion, as it was correlated with all nutrient variables in the regression model.

Study 3, established in 1978 in Wise County, Virginia, documented the productive potential and value of a white pine forest growing on reclaimed mined land and its response to thinning. Volume growth was projected to age 30 using a stand tables. Site index of the stand was found to be 32.3 m at base age 50 years. Thinning rapidly increased the diameter growth of the residual trees to 0.84cm yr⁻¹ compared to 0.58cm yr⁻¹ for the un-thinned treatment; however, at age 26, there was no difference in volume or value per hectare. At age 30, the un-thinned treatment had a volume of 457.1m³ ha⁻¹ but was only worth \$8807 ha⁻¹, while the thinned treatment was projected to have 465.8 m³ ha⁻¹, which was worth \$11265 ha⁻¹ due to a larger percentage of the volume in sawtimber size classes. These results indicate that commercial forestry is a viable alternative for reclamation of surface mined lands and that stands growing on reclaimed mined land can respond well to intermediate stand treatments.

Task 3, Sub-study 1

Experimental

Study Design. The study used a 3 x 3 factorial combination of treatments across three sites in a randomized complete block design to investigate the effects of silvicultural treatment, species assemblage, and site conditions on seedling survival and growth. This design was replicated with three blocks at each of the three study sites. The three levels of silvicultural treatment were:

1. Low intensity – weed control only (WC);
2. Medium intensity – weed control plus tillage to alleviate soil compaction (WC+T); and
3. High intensity – weed control and tillage plus fertilization to amend soil chemical properties (WC+T+F).

Species assemblages used were:

1. White pine (WP);
2. Hybrid poplar (*Populus trichocarpa* L. (Torr. & Gray ex Hook.) x *Populus deltoides* (Bartr. ex Marsh.) hybrid 52-225) (HP), and;
3. Native hardwood mix (HW).

Tree spacing was fixed for all species at 2.4 m x 3.0 m, giving a final planting density of 1,345 trees/ha. White pine and hybrid poplar were planted in pure stands, while the hardwood species mix varied at each site in order to approximate a pre-mining forest condition found in adjacent undisturbed forest (Table 3-1-1). In addition to the commercial hardwood, a combination of three nurse tree species was planted to provide early wildlife habitat and to more closely resemble the native hardwood mixture (Burger and Zipper 2002).

Table 3-1-1. Species composition and percentage of each species for the mixed hardwood plots at the three study sites.

Ohio		West Virginia		Virginia	
Species	%	Species	%	Species	%
Common Hardwoods (HW1):					
northern red oak	9.6	northern red oak	15.3	northern red oak	10.9
tulip poplar	9.6	tulip poplar	15.3	tulip poplar	10.9
sugar maple	9.6	sugar maple	15.3	sugar maple	10.9
Site-Specific Hardwoods (HW2):					
black oak	9.6	red maple	15.3	pignut hickory	10.9
chestnut oak	19.2	white ash	15.3	white oak	21.9
hickory	9.6				
scarlet oak	9.6				
Shrubs:					
redbud	7.7	redbud	7.8	redbud	7.8
flowering dogwood	7.7	flowering dogwood	7.8	flowering dogwood	7.8
Wash. hawthorn	7.7	Wash. hawthorn	7.8	Wash. hawthorn	7.8

Study sites were located in Lawrence County, Ohio, Nicholas County, West Virginia, and Wise County, Virginia, on land surface mined for coal and subsequently reclaimed according to SMCRA regulations (Fig. 3-1-1). The post-mining land use at all sites was hayland pasture, and the sites supported a dense vegetative cover composed of grasses and legumes prior to study establishment.

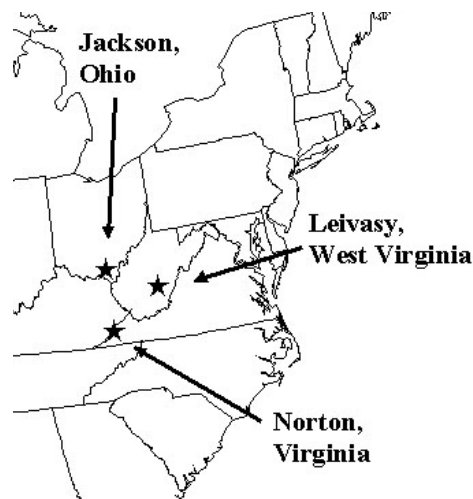


Figure 3-1-1. Location of study sites.

The site in Lawrence County, Ohio, had topsoil replaced to varying depths across the site. Both the surface soil and the subsurface soil had fine textures and low coarse fragment percentages compared to the other two sites (Tables 3-1-2 and 3-1-3). The oxidized surface soil had a much lower pH than the unoxidized subsurface material that was derived from siltstone material. The site has been reclaimed for at least 10 years and had a dense cover of predominantly tall fescue and sericea lespedeza (*Lespedeza cuneata*).

The Nicholas County, West Virginia, site did not have topsoil replaced and the material at this site was shale-derived throughout the profile. The site had coarse soil textures and high coarse fragment contents (50-60%) throughout the profile (Tables 3-1-2 and 3-1-3). The site was used for grazing prior to study establishment with the dominant grass species being tall fescue and had been reclaimed for at least 10 years.

The site in Wise County, Virginia, was derived from sandstone rocks, and soil textures ranged from loam to sandy loam. This site had an oxidized topsoil material returned to the surface throughout the plots. This site also had high coarse fragment percentages; however, the subsurface typically had more than the surface layer (Tables 3-1-2 and 3-1-3). The blocks at this site had been reclaimed for less than five years, with one block having been reclaimed the spring before study establishment. The newly reclaimed block was seeded to foxtail millet, while the other two sites were dominated by tall fescue and sweet clover.

Plots were blocked within each site based on soil properties (Tables 3-1-2 and 3-1-3). Nine 0.25-ha plots were established in each of the three blocks at each site. Plots were laid out to be as contiguous as possible within each block, while still maintaining uniform soil properties. Slopes in all plots were less than 15%.

The weed control treatment used herbicide to reduce existing herbaceous vegetation. In August 2003 a broadcast treatment of glyphosate herbicide was applied at a rate of 9.35 l ha⁻¹. Following the glyphosate treatment, a pre-emergent herbicide containing pendimethalin was applied after tree planting in April 2004 at a rate of 4.92 l ha⁻¹ to control germinating grasses. Spot applications of glyphosate were applied around each seedling in July 2004 to control competition at all study blocks, except for one block at the Virginia site where no competition was present. Seedlings were shielded from herbicide drift during application.

The tillage treatment employed was ripping. The tillage treatments varied by site depending on local equipment availability; however, the same equipment was used within individual blocks. Variations in the tillage treatment included single shank only, single shank with coulters creating beds, and multiple shanks resulting in tillage of the entire plot. The rips were spaced approximately 3 m apart and the depth of ripping was set between 61 and 91 cm. The plots were treated prior to planting in April 2004.

Fertilizer was applied to the designated plots in late May 2004. A banded application of 272 kg ha⁻¹ of diammonium phosphate added 49.0 kg ha⁻¹ N and 55.1 kg ha⁻¹ P. Muriate of potash and the micronutrient mix were applied around the base of each seedling at the following rates: 91 kg ha⁻¹ of muriate of potash added 46.8 kg ha⁻¹ K; and 20 kg ha⁻¹ of a micronutrient mix added 1.8 kg ha⁻¹ S, 0.2 kg ha⁻¹ B, 0.2 kg ha⁻¹ Cu, 0.8 kg ha⁻¹ Mn, and 4.0 kg ha⁻¹ Zn.

Table 3-1-2. Soil properties for 0-10 cm depth of study blocks at research sites in Lawrence County, OH, Nicholas County, WV, and Wise County, VA.

Site	Block	pH	Electrical Conductivity (dS m ⁻¹)	CEC (cmol _c kg ⁻¹ soil)	NaHCO ₃ Extractable P (mg kg ⁻¹)	Total N (%)	Coarse Fragments (%)	Sand- stone (%)	Silt- stone (%)	Shale (%)	Texture	Bulk Density (g cm ⁻³)
OH [†]	1	4.89	0.06	9.26	10.3	0.125	6.4	14.44	85.56	0.00	L*	1.53
OH [†]	2	5.19	0.11	7.69	7.69	0.114	6.96	27.22	61.67	0.00	L	1.44
OH [†]	3	6.05	0.13	9.05	5.38	0.106	9.86	27.22	72.78	0.00	L	1.4
Mean:		5.38	0.10	8.67	7.79	0.115	7.74	22.96	73.34	0.00	L	1.46
VA [†]	1	4.75	0.18	5.46	9.98	0.058	32.36	72.78	15.00	0.00	L	1.48
VA [†]	2	6.3	0.28	6.57	10.07	0.091	41.06	46.67	31.11	0.00	L	1.87
VA [†]	3	6.43	0.38	5.21	13.75	0.053	51.65	65.00	35.00	0.00	L/SL	1.76
Mean:		5.83	0.28	5.75	11.27	0.067	41.69	61.48	27.04	0.00	L	1.70
WV ^{††}	1	5.91	0.21	8.81	20.13	0.278	54.29	9.44	13.89	76.67	SL	1.66
WV ^{††}	2	5.72	0.22	8.37	20.81	0.258	55.26	7.22	11.67	81.11	SL	1.71
WV ^{††}	3	5.52	0.21	7.85	18.03	0.281	46.21	10.56	15.00	73.33	SL	1.67
Mean:		5.72	0.21	8.34	19.66	0.272	51.92	9.07	13.52	77.04	SL	1.68

[†] Topsoils in OH and VA were comprised of oxidized material replaced specifically as topsoil or topsoil substitute.

^{††} Topsoil in WV was the upper 10cm of soil is unoxidized and is the same material that comprises the subsoil layer.

* L = loam; SL = sandy loam

Table 3-1-3. Soil properties for subsurface samples of study blocks at research sites in Lawrence County, OH, Nicholas County, WV, and Wise County, VA.

Site	Block	pH	Electrical Conductivity (dSm ⁻¹)	CEC (cmol _c kg ⁻¹ soil)	NaHCO ₃ Extractable P (mgkg ⁻¹)	Total N (%)	Coarse Fragments (%)	Sand- stone (%)	Silt- stone (%)	Shale (%)	Texture	Bulk Density (g cm ⁻³)
OH [†]	1	6.86	0.26	16.21	0	0.048	25.41	18.89	80	1.11	SiCL	1.7
OH [†]	2	6.15	0.61	13.12	0.84	0.052	18.01	21.67	73.89	0	L	1.73
OH [†]	3	6.91	0.53	14.08	0.32	0.043	16.36	8.89	91.11	0	SiCL	1.66
Mean:		6.64	0.47	14.47	0.39	0.048	19.93	16.48	81.67	0.37	SiCL	1.70
VA [†]	1	6.77	0.21	6.02	3.38	0.060	50.27	81.43	18.57	0	SL	1.74
VA [†]	2	7.55	0.28	7.46	2.94	0.087	63.25	20	68.89	0	SL	-
VA [†]	3	6.37	0.26	4.35	2.78	0.065	56.57	66.25	33.75	0	SL	-
Mean:		6.90	0.25	5.94	3.03	0.071	56.70	55.89	40.40	0.00	SL	1.74
WV ^{††}	1	6.72	0.1	6.62	7.13	0.120	59.21	10.56	10	67.22	SL	-
WV ^{††}	2	6.03	0.12	5.89	5.94	0.101	61.56	6.67	12.22	73.11	SL	-
WV ^{††}	3	5.87	0.1	5.85	3.68	0.100	53	12.22	17.78	59	L/SL	-
Mean:		6.21	0.11	6.12	5.58	0.107	57.92	9.82	13.33	66.44	SL	-

[†] Subsurface samples in OH and VA were collected from spoil material located directly below the oxidized material at the surface that was of variable thickness.

^{††} Subsurface samples in WV were collected from 10 to 30 cm of depth as this layer was the same material that comprised the topsoil layer.

* L = loam; SiCL = silty clay loam; SL = sandy loam

Survival and Growth Data Collection. A 20 x 20 m measurement plot was established in the center of each treatment plot, within which all trees were assessed for survival, height growth, and groundline diameter growth. Initial height and groundline diameter were assessed in May 2004, shortly after bud break. First-year survival and growth was determined following measurement in late August of 2004.

Hybrid Poplar Biomass Measurements. Detailed destructive sampling to determine above- and below-ground biomass allocation was conducted in the hybrid poplar plots at the site in Nicholas County, West Virginia. Three randomly selected trees from each plot were harvested in mid-September for plant biomass determinations. Harvested trees were located outside the measurement plots used to assess survival and growth. Trees were cut off at the groundline and leaves were separated from the stems. The entire root system of each tree was carefully excavated from the soil and washed gently with water to remove soil adhering to the roots. Roots were stored at 4°C in sealed plastic bags with a moist paper towel for a period of up to four weeks, during which time the roots were separated into coarse (>0.5 mm) and fine (<0.5 mm) root fractions. All tissue samples were dried at 65°C for a minimum of 72 hours and weighed. A subsample was then ground using a Wiley mill to pass a 1-mm screen. In some instances when samples were small, a coffee grinder was used to grind all the foliage collected.

Hybrid Poplar Tissue Analysis. Tissue samples from the harvested trees in each plot were composited by the following tissue types for nutrient analysis: (1) foliage, (2) stem, and (3) roots. Total C and N were determined using an Elementar varioMAX CNS analyzer (Mt. Laurel, NJ). After dry ashing and digesting with 6N HCl, the tissue samples were analyzed using a SpectroFlame Modula Tabletop inductively coupled plasma spectrophotometer to determine elemental concentrations of P, Mg, Ca, and K for all tissue samples and S, B, Cu, Mn, and Zn for foliage samples only.

Hybrid Poplar Moisture Stress Measurements. Seedling water potential was measured using a pressure chamber (PMS Instrument Co. Model 1000 Corvallis, OR) for four consecutive rain-free days (August 16-19, 2004), with the initial measurement having been made the day after a significant rain event. Three trees from each hybrid poplar plot were measured to obtain average water potential for that plot. Measurements were timed so as to measure the water potential of all trees within a plot at the same time during the afternoon (2:30 to 6:30 p.m.) over the course of the four-day period. Water potential readings were taken immediately after the leaf was excised from the tree.

Soil samples of the surface 30 cm were collected each day from three random locations in each plot. Soil samples were stored in a sealed plastic bag and returned to the lab for determination of gravimetric soil moisture content. Soil sampling preceded water potential sampling and was confined to a time period between 12:30 and 2:00 p.m. Individual plots were sampled at the same time each of the three days.

Data Analysis. Analysis of variance was used to analyze survival and growth data for differences in survival percentage, height growth, total height, diameter growth, total diameter, volume growth, and total volume as a 3 x 3 x 3 factor random complete block design having three species assemblages, three sites, and three treatments. Tree volume was calculated as diameter squared multiplied by tree height. Analysis of variance was done by site if interaction terms containing site were significant. Additionally, if the species x treatment interaction was significant after analyzing by site, analysis of variance was done by species and by treatment to

perform mean separation procedures. For all survival and growth data, survival percentages for seedlings were transformed using the arcsine transformation and any of the growth measures that showed non-normality or heteroscedasticity were transformed using the natural log function prior to analysis of variance and subsequent mean separation procedures (Gomez and Gomez 1984).

Subsequent to the overall analysis, hardwood species were divided into three groups (Table 3-1-2) for further investigation of differences between the species used. The HW1 and shrub groups were found at each site, and the HW2 group varied by site. These data were analyzed in the same manner as the overall analysis, except that the three groups of hardwoods replaced the three species assemblages used in the overall study.

Hybrid poplar biomass data were analyzed for differences between biomass measures by treatment. Arcsine transformation was used to transform percentage data prior to analysis of variance, and any non-normal or heteroscedastic data were transformed using either the inverse or natural logarithm transformation (Gomez and Gomez 1984). Similarly, data from tissue samples were analyzed for differences between nutrient concentrations by tissue type, and non-normal and heteroscedastic data were transformed using the inverse function prior to analysis of variance. Moisture stress data was analyzed as a split-plot design with treatment as the whole plot and date as the split plots for differences between dates and treatments for soil moisture as well as plant water potential.

Mean separation was done using Tukey's HSD with significance set at $P < 0.05$ for all comparisons. If interaction terms were not significant, only main effect means were compared. SAS version 8.2 (SAS Institute Inc., Cary, NC 2001) was used for all statistical analyses.

Results

Survival. Site, species, and treatment effects, as well as both two-way interactions involving site, were all statistically significant for survival (Table 3-1-4). Treatment effects by site show that in Ohio (OH), weed control plus tillage plus fertilization (WC+T+F) significantly decreased survival to 14% compared to weed control plus tillage (WC+T) and weed control only (WC) at 49% and 51%, respectively (Table 3-1-5). At the West Virginia (WV) site, WC+T resulted in significantly higher survival than in either of the other treatments. The treatments had no effect on survival in Virginia (VA). The mean survival across species was notably higher in VA than at the other sites, though site means were not separated.

In terms of species effects, hardwood (HW) species had the highest mean survival at all sites (Table 3-1-5). This difference was significantly higher than all other species means at all other sites with the exception of hybrid poplar (HP) in OH. White pine (WP) had very low survival in OH and WV across treatments (27% and 41% respectively). Survival for HP was also low in OH and WV at 37 and 41% respectively whereas VA again had notably higher mean survivals for all species across treatments (Table 3-1-5).

Table 3-1-4. Analysis of variance results for survival and growth parameters for research sites in Lawrence County, OH, Nicholas County, WV, and Wise County, VA.

Source	Degrees of Freedom	Variable (Pr>F)						
		Survival	Height Growth	Diameter Growth	Volume Growth	Total Height	Total Diameter	Total Volume
Block	2	0.0057	0.0076	0.0812	0.8254	0.0231	0.1650	0.0270
Site	2	<0.0001	<0.0001	0.0004	<0.0001	0.0005	0.0001	0.0001
Treatment	2	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
Site*Treatment	4	0.0097	0.0003	0.0036	<0.0001	0.0535	0.0031	0.0143
Species	2	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
Site*Species	4	0.0332	<0.0001	0.0009	<0.0001	0.0021	0.0049	0.0179
Treatment*Species	4	0.3567	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
Site*Treatment*Species	8	0.3367	<0.0001	0.1146	<0.0001	0.1214	0.0678	0.3442
Model	28	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
Error*	51(52)							
Total*	79(80)							

* Degrees of freedom in parentheses are for survival only. Zero survival in one study block caused the loss of one degree of freedom from all growth variables.

Table 3-1-5. Survival percentage for research sites in Lawrence County, OH, Nicholas County, WV, and Wise County, VA by treatment and species group.

Site and Treatment	Species Group			Treatment Mean
	HW	HP	WP	
Ohio:				
WC	60	49	45	51 x **
WC+T	72	45	29	49 x
WC+T+F	18	17	6	14 y
Species Mean	50 a *	37 ab	27 b	38
West Virginia:				
WC	78	32	41	51 x
WC+T	94	62	50	69 y
WC+T+F	68	27	33	43 x
Species Mean	80 a	41 b	41 b	54
Virginia:				
WC	81	79	53	71 x
WC+T	90	70	70	77 x
WC+T+F	84	67	50	67 x
Species Mean	85 a	72 b	58 b	72

* a, b, c – For each site, values within rows with the same letter are not significantly different at $P < 0.05$.

** x, y, z – For each site, values within columns with the same letter are not significantly different at $P < 0.05$.

Height Growth. All model terms, including the three-way interaction among site, species, and treatment, were significant for height growth (Table 3-1-4). Analysis by site revealed that the species by treatment interaction was not significant in OH and that there were no treatment effects at this site (Table 3-1-6). Species by treatment interaction was still significant in WV and VA when analyzed by site. Significant treatment effects in WV included higher mean height growth for HW in WC+T+F than in WC as well as lower mean height growth for HP in WC (22.4 cm) compared to both of the other treatments (60.2 cm for WC+T and 57.6 cm for WC+T+F). In VA, both HW and HP had significantly more height growth as a result of WC+T+F than in either of the other treatments, with this difference being pronounced for HP (126.6 cm in WC+T+F versus 40.9 and 65.4 cm for WC and WC+T, respectively) (Table 3-1-6).

Considering species effects, HP grew significantly more than HW and WP across treatments in OH (45.6cm versus -2.3cm and 6.0cm respectively) and HW had negative height growth means for all treatments at this site (Table 3-1-6). In WV, HP grew significantly more than HW and WP in all treatments. In the WC treatment, WP also had significantly more height growth than HW (5.5cm versus -1.4cm). At the VA site, HP grew significantly more than HW and WP (Table 3-1-6).

Table 3-1-6. Average height growth (cm) for research sites in Lawrence County, OH, Nicholas County, WV, and Wise County, VA, by treatment and species group.

Site and Treatment	Species Group			Treatment Mean
	HW	HP	WP	
Ohio:				
WC	-1.0	35.8	5.2	13.3 x**
WC+T	-3.7	50.3	5.4	17.4 x
WC+T+F	-2.3	50.8	7.9	20.2 x
Species Mean	-2.3 a*	45.6 b	6.0 a	16.8
West Virginia:				
WC	-1.4 a x	22.4 b x	5.5 a x	8.8
WC+T	3.2 a xy	60.2 b y	8.9 a x	24.1
WC+T+F	7.7 a y	57.6 b y	5.8 a x	23.7
Species Mean	3.2	46.7	6.7	18.9
Virginia:				
WC	3.7 a x	40.9 b x	6.0 a x	16.9
WC+T	3.9 a x	65.4 b x	5.9 a x	25.1
WC+T+F	7.9 a y	126.6 b y	5.5 a x	46.7
Species Mean	5.2	77.6	5.8	29.5

* a, b, c – For each site, values within rows with the same letter are not significantly different at $P < 0.05$.

** x, y, z – For each site, values within columns with the same letter are not significantly different at $P < 0.05$.

Total Height. All model terms except the three-way interaction among site, species, and treatment and the site by treatment interaction were significant for total height after one growing season (Table 3-1-4). Analysis by site revealed that the species by treatment interaction was not significant in OH and that there were no treatment effects at this site (Table 3-1-7). In WV and VA, species interacted significantly with treatment. Treatment effects by species in WV included larger final heights for HP in WC+T and WC+T+F compared to WC and more total height for WP in WC+T compared to the other treatments. In VA, the only significant treatment response was a higher final height for HP in WC+T+F than the other two treatments (Table 3-1-7).

As expected, species effects for total height were somewhat different than those for height growth. In OH, mean HW height across treatments was 30.1 cm, which was not significantly different from HP or WP at 45.6 and 23.3 cm, respectively. HP and WP total heights were significantly different from each other at this site (Table 3-1-7). In WV, all three species were significantly different at each level of treatment with the species by treatment interaction occurring in the WC treatment where HP was found to have the shortest total height as opposed to the other two treatments where HP had the tallest total height (Table 3-1-7). In VA, HW species were not significantly different from WP or HP, which were significantly different from each other for WC. In WC+T and WC+T+F at this site, all species were significantly different. Mean WP heights were the shortest and HP heights the tallest for all treatments at this site (Table 3-1-7).

Table 3-1-7. Total tree height (cm) for research sites in Lawrence County, OH, Nicholas County, WV, and Wise County, VA, by treatment and species group.

Site and Treatment	Species Group			Treatment Mean
	HW	HP	WP	
Ohio:				
WC	30.8	35.8	22.4	29.7 x**
WC+T	25.0	50.3	24.7	33.4 x
WC+T+F	34.4	50.8	22.6	37.6 x
Species Mean	30.1 ab*	45.6 a	23.3 b	33.4
West Virginia:				
WC	32.4 a x	22.4 b x	25.2 c x	26.6
WC+T	38.6 a x	60.2 b y	28.2 c y	42.3
WC+T+F	36.5 a x	57.6 b y	22.9 c x	39.0
Species Mean	35.8	46.7	25.4	36.0
Virginia:				
WC	33.1 ab x	40.9 a x	23.5 b x	32.5
WC+T	37.0 a x	65.4 b x	25.0 c x	42.4
WC+T+F	40.6 a x	126.6 b y	22.6 c x	63.3
Species Mean	36.9	77.6	23.7	46.1

* a, b, c – For each site, values within rows with the same letter are not significantly different at $P < 0.05$.

** x, y, z – For each site, values within columns with the same letter are not significantly different at $P < 0.05$.

Diameter Growth. All model terms were significant for diameter growth with the exception of the three-way interaction among site, species, and treatment (Table 3-1-4). Analysis by site eliminated the species by treatment interaction at the site in OH and showed that there were no treatment effects at this site on diameter growth. The species by treatment interaction in WV was likely due to the significant diameter growth response to WC+T (7.0mm) and WC+T+F (7.5 mm) compared to WC (3.1 mm) when no other species had a significant response to treatment at this site (Table 3-1-8). Similarly, species by treatment interaction in VA was likely due to the significant response for HP in WC+T+F (13.9 mm) compared to WC+T (7.0 mm) and WC (4.9 mm). There were no other treatment differences at this site.

Species effects show that HP had significantly more diameter growth than any other species at any level of treatment at all three sites (Table 3-1-8). In WV, where species by treatment interaction was significant, in addition to the HP response HW grew significantly more in diameter than WP. There were no other differences in species in VA other than the HP response.

Table 3-1-8. Average diameter growth (mm) for research sites in Lawrence County, OH, Nicholas County, WV, and Wise County, VA, by treatment and species group.

Site and Treatment	Species Group			Treatment Mean
	HW	HP	WP	
Ohio:				
WC	0.9	4.1	0.9	2.0 x**
WC+T	0.8	5.5	0.5	2.3 x
WC+T+F	0.3	7.4	0.7	3.1 x
Species Mean	0.7 a *	5.7 b	0.7 a	2.4
West Virginia:				
WC	0.9 a x	3.1 b x	0.5 c x	1.5
WC+T	1.4 a x	7.0 b y	0.7 a x	3.0
WC+T+F	1.8 a x	7.5 b y	0.9 a x	3.4
Species Mean	1.4	5.9	0.7	2.6
Virginia:				
WC	0.8 a x	4.9 b x	0.6 a x	2.1
WC+T	1.4 a x	7.0 b x	0.6 a x	3.0
WC+T+F	2.1 a x	13.9 b y	0.7 a x	5.6
Species Mean	1.4	8.6	0.6	3.6

* a, b, c – For each site, values within rows with the same letter are not significantly different at $P < 0.05$.

** x, y, z – For each site, values within columns with the same letter are not significantly different at $P < 0.05$.

Total Diameter. As with diameter growth, all model terms were significant for total diameter with the exception of the three-way interaction between site, species, and treatment (Table 3-1-4). Analysis by site eliminated the species by treatment interaction at the site in OH and showed that there were no treatment effects at this site on total diameter (Table 3-1-9). In WV and VA, species and treatment interacted significantly. There were no significant treatment effects in WV for any species, while in VA; the only significant treatment response was for HP, where the WC+T+F total diameter (13.6 mm) compared to the WC+T (7.0 mm) and WC (4.9 mm).

There were no significant differences among species in OH across all treatments (Table 3-1-9). The significant interaction between species and treatment in WV was attributable to HP in WC being significantly less than HW and WP, whereas HP is no different than HW in WC+T and significantly higher than WP in WC+T and HW and WP in WC+T+F. There were no species differences in WC at the site in VA (Table 3-1-9), where species by treatment interaction was significant. For WC+T, HP was significantly greater than WP, and for WC+T+F, HP was significantly greater than both WP and HW.

Table 3-1-9. Total tree diameter (mm) for research sites in Lawrence County, OH, Nicholas County, WV, and Wise County, VA, by treatment and species group.

Site and Treatment	Species Group			Treatment Mean
	HW	HP	WP	
Ohio:				
WC	5.3	4.1	4.6	4.7 x**
WC+T	4.5	5.5	4.3	4.8 x
WC+T+F	4.9	7.4	3.8	5.6 x
Species Mean	4.9 a*	5.7 a	4.3 a	5.0
West Virginia:				
WC	5.2 a x	3.1 b x	4.5 a x	4.3
WC+T	5.6 ab x	7.0 a x	5.1 b x	5.9
WC+T+F	5.7 a x	7.5 a x	4.3 a x	5.8
Species Mean	5.5	5.9	4.7	5.3
Virginia:				
WC	4.9 a x	4.9 a x	5.0 a x	4.9
WC+T	5.6 ab x	7.0 a x	4.9 b x	5.8
WC+T+F	6.5 a x	13.9 b y	4.8 a x	8.4
Species Mean	5.7	8.6	4.9	6.4

* a, b, c – For each site, values within rows with the same letter are not significantly different at $P < 0.05$.

** x, y, z – For each site, values within columns with the same letter are not significantly different at $P < 0.05$.

Volume Growth. For volume growth, all model terms, including the three-way interaction among site, species, and treatment, were significant (Table 3-1-4). Analysis by site eliminated the species by treatment interaction in OH and showed no differences among treatments for volume growth at this site (Table 3-1-10). Species by treatment interaction was significant for WV and VA when analyzed by site. The interaction in WV was found to be the significant volume growth response in WC+T (4.2 cm³) for WP compared to WC (2.3 cm³), though neither treatment was significantly different from WC+T+F (2.8 cm³). There was also a notable response for HP to WC+T (43.3 cm³) and WC+T+F (51.7 cm³) compared to WC (2.8 cm³). In VA, the interaction is explained by the positive response to WC+T+F for HW and HP, while WP was unresponsive. The response was significant compared to WC and WC+T for HP and compared only to WC for HW (Table 3-1-10).

Looking at species effects in OH, HP had significantly more volume growth than HW or WP across treatments (Table 3-1-10). The species differences in WV occurred in WC+T, where HP had significantly more volume growth than HW or WP, and in WC+T+F, where HP volume growth was significantly higher than that for WP. In VA, HP volume growth was significantly higher than HW and WP for all treatments (Table 3-1-10).

Table 3-1-10. Average volume growth (cm³) for research sites in Lawrence County, OH, Nicholas County, WV, and Wise County, VA, by treatment and species group.

Site and Treatment	Species Group			Treatment Mean
	HW	HP	WP	
Ohio:				
WC	4.1	11.3	2.8	6.1 x**
WC+T	2.0	26.2	1.9	10.0 x
WC+T+F	0.2	54.5	2.1	21.0 x
Species Mean	2.1 a *	30.7 b	2.3 a	12.4
West Virginia:				
WC	2.3 a x	2.8 a x	2.3 a x	2.4
WC+T	7.3 a x	43.3 b y	4.2 a y	18.2
WC+T+F	10.4 ab x	51.7 a y	2.8 b xy	21.6
Species Mean	6.6	32.6	3.1	14.1
Virginia:				
WC	4.0 a x	15.6 b x	2.8 a x	7.5
WC+T	8.2 a xy	54.1 b x	2.7 a x	21.7
WC+T+F	16.5 a y	312.1 b y	2.7 a x	110.4
Species Mean	9.5	127.3	2.8	46.5

* a, b, c – For each site, values within rows with the same letter are not significantly different at $P < 0.05$.

** x, y, z – For each site, values within columns with the same letter are not significantly different at $P < 0.05$.

Total Volume. All model terms with the exception of the three-way interaction among site, species, and treatment were significant for total volume (Table 3-1-4). Analysis by site eliminated the species by treatment interaction for OH, where no treatment effects were evident (Table 3-1-11). Species by treatment interaction was significant for WV and VA. The only treatment effect in WV was a significantly higher total volume in WC+T and WC+T+F compared to WC only for HP. In VA, HW had significantly more volume in WC+T+F than WC, while for HP, WC+T+F had significantly more volume (312.1 cm³) than both WC+T and WC (54.1 and 15.6 cm³, respectively) (Table 3-1-11).

In terms of species differences, HP had significantly more final volume than WP in OH. In WV, HP volume was the lowest and HW the highest in WC, where all three species were significantly different (Table 3-1-11). HP was significantly higher than all other species in WC+T and WP was significantly lower than the other species in WC+T+F in WV. In VA, there were no differences in volume among species in WC. In WC+T, HP had significantly more volume than HW or WP, and in WC+T+F, all three species were significantly different, with HP having the highest volume (312.1 cm³) followed by HW (25.9 cm³) and WP (6.2 cm³).

Table 3-1-11. Total tree volume (cm³) for research sites in Lawrence County, OH, Nicholas County, WV, and Wise County, VA, by treatment and species group.

Site and Treatment	Species Group			Treatment Mean
	HW	HP	WP	
Ohio:				
WC	13.9	11.3	5.6	10.3 x**
WC+T	7.4	26.2	5.6	13.1 x
WC+T+F	14.7	54.5	3.9	26.9 x
Species Mean	12.0 ab*	30.7 a	5.2 b	16.4
West Virginia:				
WC	11.7 a x	2.8 b x	6.2 c x	6.9
WC+T	17.6 a x	43.3 b y	9.0 a x	23.3
WC+T+F	16.4 a x	51.7 a y	5.2 b x	24.4
Species Mean	15.2	32.6	6.8	18.2
Virginia:				
WC	11.4 a x	15.6 a x	6.9 a x	11.3
WC+T	17.6 a xy	54.1 b x	7.0 a x	26.3
WC+T+F	25.9 a y	312.1 b y	6.2 c x	114.7
Species Mean	18.3	127.3	6.7	50.8

* a, b, c – For each site, values within rows with the same letter are not significantly different at $P < 0.05$.

** x, y, z – For each site, values within columns with the same letter are not significantly different at $P < 0.05$.

Hardwood Species Differences

Survival. For hardwood survival percentage, site, treatment, species group, and the site by treatment interaction terms were significant in the model (Table 3-1-12). Analysis of treatment effects by site revealed that WC+T+F significantly decreased survival across species in OH to 16% compared to WC and WC+T, which had survival percentages of 60% and 71%, respectively (Table 3-1-13). In WV, WC+T+F decreased survival (63%) significantly compared to WC+T (94%). There were no differences in survival among treatments in VA.

The survival of the site-specific hardwood species (HW2) (Table 3-1) was significantly higher than either the HW1 group consisting of red oak, sugar maple (*Acer saccharum* Marsh.), and tulip poplar, or the shrub group (Table 3-1-13).

Table 3-1-12. Analysis of variance results for survival and growth parameters for hardwood groups at research sites in Lawrence County, OH, Nicholas County, WV, and Wise County, VA.

Source	Degrees of Freedom	Variable (Pr>F)		
		Survival	Height Growth	Total Height
Block	2	<0.0001	0.0045	0.0682
Site	2	<0.0001	<0.0001	0.0001
Treatment	2	<0.0001	<0.0001	0.2632
Site*Treatment	4	0.0105	0.0429	0.1458
Species	2	0.0005	<0.0001	<0.0001
Site*Species	4	0.9222	0.0087	0.1914
Treatment*Species	4	0.8364	0.1485	0.1996
Site*Treatment*Species	7	0.5439	0.3193	0.9807
Model	27(28)	<0.0001	<0.0001	0.0003
Error*	50(51)			
Total*	77(79)			

* Degrees of freedom in parentheses are for survival only. Zero survival for shrubs in three study blocks caused the loss of one degree of freedom from all growth variables.

Table 3-1-13. Survival percentage of hardwood species groups for research sites in Lawrence County, OH, Nicholas County, WV, and Wise County, VA, by treatment and species group.

Site and Treatment	Species Group			Treatment Mean
	HW1	HW2	Shrub	
Ohio:				
WC	66	67	49	60 x **
WC+T	64	82	65	71 x
WC+T+F	15	27	0	16 y
Species Mean	48 a *	59 b	43 a	50
West Virginia:				
WC	71	87	81	80 xy
WC+T	92	96	93	94 x
WC+T+F	68	86	35	63 y
Species Mean	77 a	90 b	69 a	79
Virginia:				
WC	82	92	69	81 x
WC+T	86	96	89	90 x
WC+T+F	79	89	79	82 x
Species Mean	82 a	92 b	79 a	85

* a, b, c – For each site, values within rows with the same letter are not significantly different at $P < 0.05$.

** x, y, z – For each site, values within columns with the same letter are not significantly different at $P < 0.05$.

Height Growth. Site, species group, treatment, and the site by treatment and site by species interactions were significant terms in the model for height growth of hardwoods (Table 3-1-12). Analysis by site revealed that there were no treatment effects in OH across species groups, and all treatment means were negative (Table 3-1-14). In WV, both WC+T and WC+T+F were significantly higher than WC (3.6 cm and 7.4 cm, respectively, versus -1.2 cm). In VA, WC+T+F had a mean height growth of 9.8 cm, which was significantly higher than the 4.0 cm resulting from WC and the 4.4 cm resulting from WC+T.

The HW1 species had the lowest mean height growth at all sites (Table 3-1-14). In OH, the height growth of -6.3 cm was significantly less than the mean height growth for shrubs of 1.5 cm. Height growth of HW1 species was significantly less than both HW2 and shrub groups in WV, and this mean value was negative (-1.1 cm), whereas the other two groups had positive mean height growth values (6.9 and 4.1 cm, respectively). HW1 species in VA were no different from HW2 species, which both had positive mean height growth at this site. Both of these groups had significantly less height growth than the shrub group in VA (Table 3-1-14).

Table 3-1-14. Height growth (cm) of hardwood species groups for research sites in Lawrence County, OH, Nicholas County, WV, and Wise County, VA, by treatment and species group.

Site and Treatment	Species Group			Treatment Mean
	HW1	HW2	Shrub	
Ohio:				
WC	-1.9	-1.0	0.5	-0.8 x **
WC+T	-13.5	-3.2	2.5	-4.7 x
WC+T+F	-3.6	-2.1	---	-2.9 x
Species Mean	-6.3 a *	-2.1 ab	1.5 b	-2.8
West Virginia:				
WC	-5.9	3.0	-0.8	-1.2 x
WC+T	0.0	6.9	4.0	3.6 y
WC+T+F	2.5	10.8	9.1	7.4 y
Species Mean	-1.1 a	6.9 b	4.1 b	3.3
Virginia:				
WC	1.5	2.7	7.9	4.0 x
WC+T	0.2	3.8	9.2	4.4 x
WC+T+F	2.3	7.4	19.7	9.8 y
Species Mean	1.3 a	4.6 a	12.2 b	6.1

* a, b, c – For each site, values within rows with the same letter are not significantly different at $P < 0.05$.

** x, y, z – For each site, values within columns with the same letter are not significantly different at $P < 0.05$.

Total Height. Site and species were the only significant terms in the model for total height (Table 3-1-12). There were no treatment effects for any site or species (Table 3-1-15). Examining species differences in total height across sites and treatments revealed that HW2 species were significantly shorter than HW1 or shrub species. Lack of interaction in this analysis facilitated the comparison of site main effects, where WV and VA were found to have significantly greater total heights than OH (Table 3-1-15).

Table 3-1-15. Total tree height (cm) of hardwood species groups for research sites in Lawrence County, OH, Nicholas County, WV, and Wise County, VA, by treatment and species group.

Site and Treatment	Species Group			Treatment Mean
	HW1	HW2	Shrub	
Ohio:				
WC	39.1	24.3	28.8	30.7 x ^{**}
WC+T	27.1	20.8	31.0	26.3 x
WC+T+F	42.2	25.3	---	33.7 x
Species Mean	36.1 a [*]	23.5 b	29.9 a	29.8 m [†]
West Virginia:				
WC	34.2	30.3	32.6	32.4 x
WC+T	40.2	32.3	45.1	39.2 x
WC+T+F	38.7	34.7	34.9	36.1 x
Species Mean	37.7 a	32.4 b	37.5 a	35.9 n
Virginia:				
WC	38.5	27.9	36.4	34.3 x
WC+T	37.5	27.2	52.8	39.2 x
WC+T+F	45.7	33.0	53.2	44.0 x
Species Mean	40.5 a	29.4 b	47.5 a	39.1 n

* a, b, c – For each site, values within rows with the same letter are not significantly different at $P < 0.05$.

** x, y, z – For each site, values within columns with the same letter are not significantly different at $P < 0.05$.

† m, n – For overall site means, values with the same letter are not significantly different at $P < 0.05$.

Hybrid Poplar Biomass

Total plant biomass differences increased significantly with the intensity of silvicultural input. Root, stem, and foliage biomass also increased significantly with the level of silvicultural intensity (Figure 3-1-2). The percentage of fine roots (<0.5 mm) was the same for the WC+T+F and WC+T treatments (23%), while the WC plots had a much higher fine root percentage (54%), which was significantly different from the other two treatments. Additionally, the root-to-shoot ratios were not significantly different between WC+T and WC+T+F (0.31 and 0.37, respectively), but both were significantly higher than that of the WC treatment (0.08).

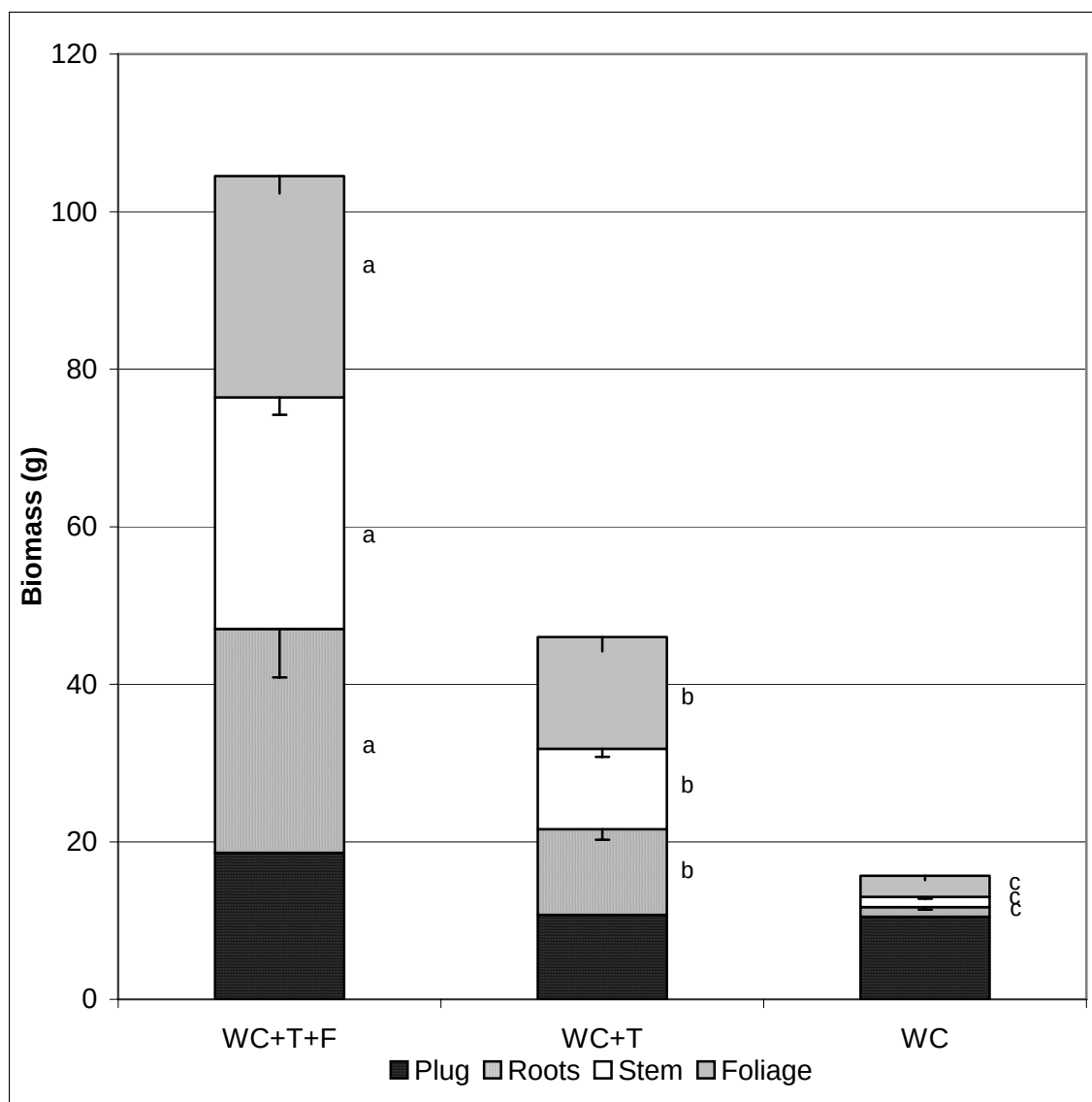


Figure 3-1-2. Hybrid poplar biomass by plant part and treatment for study site in Nicholas County, WV. Letters beside segments indicate significant differences at the $P < 0.05$ level among treatments for that particular segment.

Hybrid Poplar Tissue Analysis

Foliar nutrient concentrations were significantly higher for N, P, and Mn in the WC+T+F treatment compared to the other two treatments (Table 3-1-16). Foliar K in the WC+T+F treatment was only significantly higher than in the WC+T treatment. There were no differences between treatments for any other nutrients.

For stem tissue, N was the only added nutrient that had a higher mean concentration in the WC+T+F treatment and this mean was only significantly different from the WC treatment (Table 3-1-16). The concentration of N in the root tissue was significantly higher for the WC+T+F treatment compared to the WC+T treatment, but was not significantly different from the WC treatment.

Table 3-1-16. Macro- and micronutrient concentrations by tissue type and treatment for hybrid poplar growing at the research site in Nicholas County, WV.

Tissue Type and Treatment	Macronutrients (g kg ⁻¹)					Micronutrients (mg kg ⁻¹)				
	N	P	K	Mg	Ca	S	Zn	B	Cu	Mn
Foliage:										
WC	24.16 x*	1.98 x	14.19 x	4.60 x	12.14 x	3.92 x	84.30 x	30.04 x	8.95 x	161.17 x
WC+T	26.09 x	1.93 x	15.89 xy	4.86 x	12.26 x	4.82 x	92.21 x	26.61 x	9.71 x	134.44 x
WC+T+F	32.58 y	2.32 y	17.28 y	5.11 x	11.95 x	4.42 x	84.94 x	46.98 x	10.92 x	309.97 y
Stem:										
WC	7.16 x	0.37 x	2.76 x	0.51 x	1.37 x					
WC+T	7.40 xy	0.24 x	2.14 x	0.51 x	1.25 x					
WC+T+F	8.33 y	0.25 x	1.71 x	0.41 x	0.98 x					
Root:										
WC	9.38 xy	1.06 x	8.68 x	1.80 x	6.30 x					
WC+T	7.88 x	0.95 x	10.65 x	1.79 x	7.28 x					
WC+T+F	11.26 y	1.21 x	10.63 x	2.01 x	7.55 x					

* For a given plant part, different letters within a column indicate significant differences at $P < 0.05$.

Hybrid Poplar Moisture Stress

There was a statistically significant decrease with each successive day for all treatments. The WC only and WC+T treatments were significantly different over all three days of the dry down period (Table 3-1-17).

The treatment by date interaction was significant for water potential means. Each treatment increased or remained the same over the first three days of the dry down experiment. No means were statistically significant for the first and third days with respect to treatments. The WC+T+F treatment was significantly different from the other treatments for day 2. For the final day, however, the WC+T treatment continued to increase rapidly (Table 3-1-17) to -2.30 MPa, which was significantly higher than the WC treatment, which had decreased to -1.62 MPa. The WC+T+F and WC treatment means decreased on day 4, likely as a result of the cloud cover present over the site that particular day.

Table 3-1-17. Gravimetric soil moisture and water potential for hybrid poplar growing at the research site in Nicholas County, West Virginia.

Treatment	Aug. 16	Aug. 17	Aug. 18	Aug. 19	Treatment Average
Gravimetric Soil Moisture (kg kg ⁻¹)					
WC	---	0.16	0.15	0.12	0.14 x ^{**}
WC+T	---	0.14	0.13	0.12	0.13 y
WC+T+F	---	0.15	0.12	0.12	0.13 xy
Date average	---	0.15 a [*]	0.13 b	0.12 c	0.13
Water Potential (MPa)					
WC	-1.30 a x	-1.66 b x	-1.89 b x	-1.62 ab x	-1.62
WC+T	-1.32 a x	-1.72 ab x	-1.90 bc x	-2.30 c y	-1.81
WC+T+F	-1.17 a x	-1.97 b y	-1.97 b x	-1.78 b xy	-1.72
Date average	-1.26	-1.78	-1.92	-1.90	-1.72

* a, b, c – Values within rows with the same letter are not significantly different at $P < 0.05$.

** x, y, z – Values within columns with the same letter are not significantly different at $P < 0.05$.

Discussion

The results of this study point to the fact that there is likely no universal prescription for good establishment and first-year growth of the species used in this study, as numerous interactions existed among the sites, treatments, and species assemblages used in this study. Survival differences in this study were largely attributable to site differences, as the overall analysis of variance showed significant site by treatment and site by species interactions (Table 3-1-4). Overall survival was lowest in OH, with white pine having the lowest survival of the three species assemblages at this site. Larson et al. (1995) found white pine to survive and grow poorly on sites in this geographic area with the near alkaline and fine texture spoil materials common to the area. At this site, WC+T+F decreased survival below that of the WC-only treatment, which did not occur at the other two sites, though mean survival was less in WC+T+F than in WC at the other two sites. Two hypotheses exist for decreased survival in WC+T+F: (1) fertilization stimulated the competing vegetation (Ramsey et al. 2001); and/or (2) a salt effect was created by the fertilizer, leading to moisture stress in the trees such as that postulated by van den Driessche and coworkers (2003) for aspen seedlings. In OH, a combination of these two hypotheses would be more likely because, despite uniform herbicide applications at all sites, OH was observed to have much more competing vegetation by the end of the growing season than either of the other sites. Additionally, the spoils at this site, though covered with an oxidized topsoil material, were still generally within the rooting zone of the trees, especially in WC+T and WC+T+F, where tillage brought this material closer to the surface. The spoil materials at this site were found to be near alkaline and to have a much higher cation exchange capacity relative to the other sites (Tables 3-1-2 and 3-1-3). Both electrical conductivity and pH were within ranges (>0.05 dS m⁻¹ and pH > 6.0 , respectively) reported by Torbert et al. (1994) as negatively affecting white pine growth. Electrical conductivity in both surface and subsurface layers at the sites in WV and VA had values less than 0.05 dS m⁻¹. In addition to this, the loam to sandy loam textures at this site could have allowed excess salts to leach faster than what would be expected in the finer-textured structureless spoils in OH.

In addition to added weed control as a result of tillage, tillage has been shown to reduce bulk density and consequently increase survival and growth of trees on reclaimed surface mines (Ashby 1996, Torbert and Burger 1996). Tillage in this study produced mixed results in terms of providing a survival and growth response. In OH, WC+T did not produce a significant response compared to WC. The tillage treatment was carried out at this site when the soils were very wet. Given that the soils at this site are fine-textured and that it was only 20% coarse fragments, which would tend to aid in loosening the soil during ripping, it is possible that the reason for the lack of response is that the tillage failed to sufficiently loosen the soil at this site. There was a notable response to WC+T for HP in WV in terms of survival, growth, and biomass measures, indicating that soil compaction is likely a limiting factor for good growth at this site. Interestingly, the site in VA had similar soil textures and bulk densities to the WV site, but did not have a significant response to WC+T; however, mean survival and mean growth for all measures across species were higher in WC+T than in WC only.

The sites in VA and WV were both characterized by coarse textured spoils and high coarse fragment contents (Tables 3-1-2 and 3-1-3). The primary differences between these two sites were rock type, topsoil replacement, and average age since reclamation. Looking at survival differences between these two sites, it can be seen that survival in VA was better than that in WV, regardless of species or treatment. This again could be a result of better weed control on the younger spoils in VA, where the seed pool for competing vegetation may have been smaller. Additionally, soil N and P levels were much higher in the surface layer in WV than in the subsurface layer at the same site and higher than the surface or subsurface layer in VA and OH. This may be evidence that the site was fertilized to maintain the lush grass cover for the grazing animals that occupied the site prior to study establishment. This could make weed control more difficult at this site, creating conditions conducive to competition growth in plots that did not receive fertilization and in the WC+T+F plots, by adding nutrients in excess of what the trees likely needed. In WV, WC+T produced the largest response in both survival and growth of HP and WP. The lack of a significant growth response due to fertilization, especially for the site-demanding poplars, may be explained by the higher levels of N and P at this site compared to the other sites. Hybrid poplar at this site did, however, have significantly higher root, stem, foliage, and total plant biomass in the fertilized plots versus the other two treatments. Fertilized trees appeared to have improved water relations compared to WC+T. Harvey and van den Driessche (1997) found that N fertilization alone increased drought resistance of *Populus trichocarpa* Torr. & Gray, but fertilization with P alone increased drought resistance and suggest that fertilization with N and P may allow good growth without leading to poor water relations. The growth response to WC+T+F in VA is reasonable given the previously mentioned considerations, namely (1) better weed control relative to the other two sites, and (2) inherently low N and P levels in both soil layers. The response to WC+T+F was exceptional for HP in VA, where total stem volumes were 312.1 cm³ and total heights were 126.6 cm. The next closest total height was also in VA in WC+T at 65.4 cm, followed by WC+T in WV at 60.2 cm. This species has been shown to be very responsive to fertilization with N in combination with P when soil fertility levels are low (van den Driessche 1999, Brown and van den Driessche 2005).

A study conducted on three surface mines in West Virginia using similar species and treatments found trends similar to those in our study (McGill et al. 2004). For instance, the same hybrid poplar clone averaged 1.0 m in total height after one year, and average survival for this same species across all three sites was found to be 79%. Additionally, these authors found

excellent survival (>90%) for the two hardwood species used and found low survival for white pine (48%) at the one site on which this species was planted.

Conclusions

Several conclusions can be drawn from the results of this investigation, including:

1. White pine and hardwood species grew little over the course of the first growing season, which could translate into a need for continued weed control to ensure the trees do not succumb to the competing vegetation as mean heights ranged from 25 to 40 cm for hardwoods and from 20 to 30 cm for white pine.
2. Hardwood species had excellent survival in WV and VA, and better survival than the other species used in OH, while white pine had the poorest survival of all species at all sites. Survival was particularly good for the site-specific hardwoods planted at each site.
3. Weed control plus tillage may be the optimum treatment for hardwoods and white pine, as any increased growth resulting from fertilization may not offset the decreased survival that accompanied fertilization.
4. Hybrid poplar appears to have good potential for reverting post-SMCRA mined lands reclaimed to grasses back to forests, as this species had good growth with 50 to 65 cm of height growth in one year in WC+T at all sites and excellent growth in WC+T+F in VA. This good growth, coupled with survival percentages that may be adequate to ensure that without further weed control, an adequately stocked stand could develop, gives this species an advantage over the other species used in this study.
5. Considering mean survival and growth, sites with soil characteristics similar to those at the site in VA would seem to be very suitable for tree survival and growth, while those in WV appear to be less suitable with the treatments and species used, and on sites similar to OH, other treatments and/or species may be needed to ensure good establishment and growth of forest stands.
6. Though height and diameter growth were not statistically different for HP for WC+T and WC+T+F in WV, biomass responded significantly to each level of silvicultural input, with WC+T+F trees also showing improved foliar nutrition compared to WC and WC+T.

Task 3, Sub-study 2

Experimental

Site Description and Classification. In a previous study, Burger et al. (2002) developed a preliminary mine soil quality classification model that included three criteria: rock type, compaction, and aspect (Fig. 3-2-1). These criteria were based on a conceptual model of mine soil quality factors that included soil strength or density, air/water balance, soil temperature, and nutrient level. The mapping criterion rock type (Fig. 3-2-1) is a field indicator of soil air/water balance and nutrient level. The level of soil compaction is an indicator of soil strength and air/water balance, while aspect is an indicator of air/water balance and soil temperature. A weighting factor (WF) was assigned to each of the three criteria to reflect estimated relative importance (Burger et al. 2002). Each sampling site was classified by assigning a value of 1 to 5 for each criterion, multiplying that value by its respective WF, and summing these products to obtain a soil/site quality class (SQC) ranging from I to V (Burger et al. 2002), expressed as Roman numerals as per traditional expression of site quality gradients for forestry. Our study was used to test the site classification model and to determine if the criteria used adequately represented the factors that influenced white oak growth.

Our study was conducted during the summer of 2004. The study area is located in Buchanan County in southwestern Virginia on a reclaimed coal mine owned by Rapoca Energy Company. This area is in the Kanawha geologic formation, and various rock strata from above the coal seam were placed on the surface during reclamation. As a result, the reclaimed area has a wide variety of mine soil types, varying in both physical and chemical properties. Reclamation grading operations were conducted with the intent of avoiding surface compaction so as to create favorable conditions for reforestation.

In 2001, immediately after final grading, reclaimed soils were mapped using the proposed model and areas of the site with similar mine-soil characteristics were delineated as soil mapping units. For each mapping unit, SQC was determined using the model described above (Fig. 3-2-1). The site was planted in 2002 with native hardwoods at a density of 1482 trees ha⁻¹ using species mixes appropriate for each mapping unit's soil properties and site quality. White oak was planted on all sites. White oak seedlings were obtained from the Virginia Department of Forestry state nursery. Those planted were of uniform grade and had a relatively uniform initial caliper and height. Herbaceous vegetation was applied uniformly over the site by a seeding contractor.

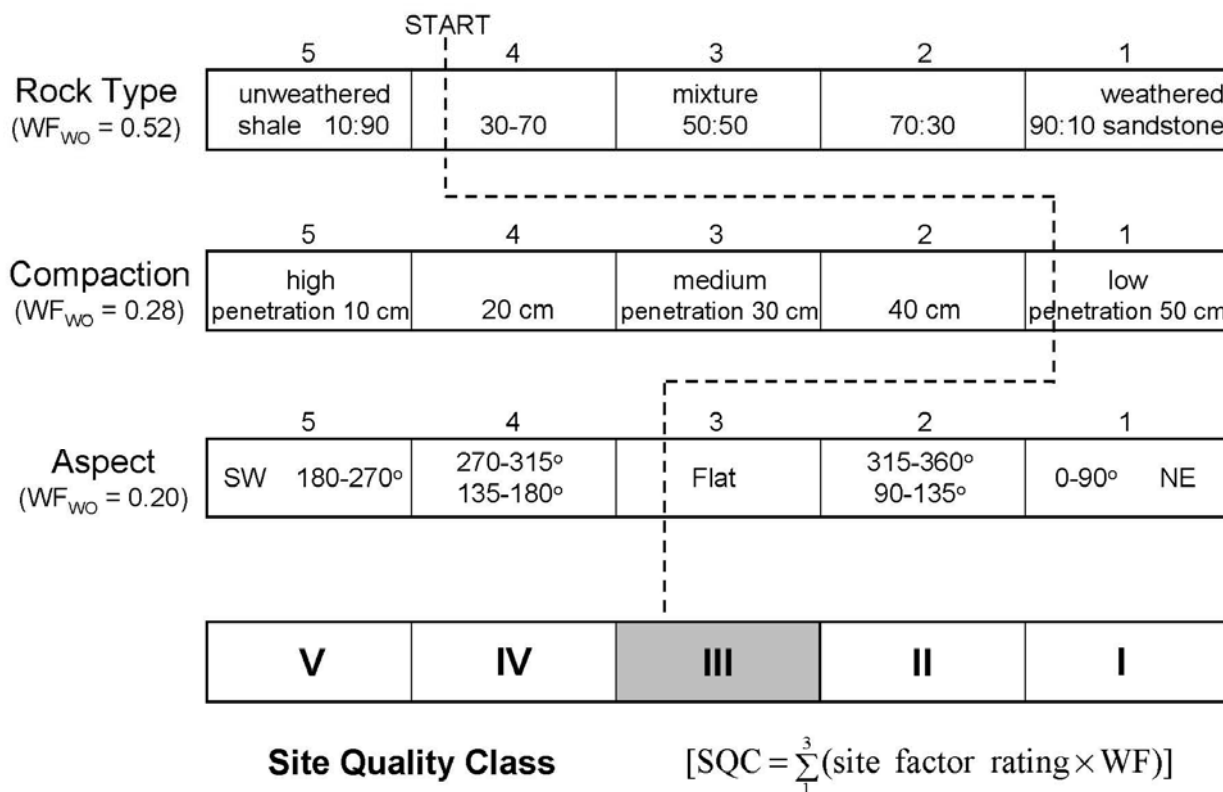


Figure 3-2-1. Site factor gradients used to determine overall mined site quality class (SQC). The field mapping criteria, rock type, level of compaction and aspect, are scored from 1 to 5 and the score is adjusted by multiplying by their respective weighting factors (WF). SQC is determined by summing the adjusted scores (Burger et al. 2002). The word “Start” shows an example culminating in a SQC III.

Field and Laboratory Methods. Total tree height and third-year growth were used to judge the accuracy of the SQC assignments. Seventy-two three-year-old white oak trees were randomly chosen across a range of spoil types, which included six mapping units (Fig. 3-2-2). White oak was chosen as the test species because it was a component of all species mixes, is a valuable timber species of the late successional native forest of the Appalachians, and is found throughout the Appalachians on a range of sites. Glyphosate was sprayed in a 1-m diameter circle around each tree at the beginning and in the middle of the growing season to minimize the effect of herbaceous competition on third-year growth. Total tree height and third-year growth were measured at the end of the third growing season, and averages were calculated for each mapping unit.

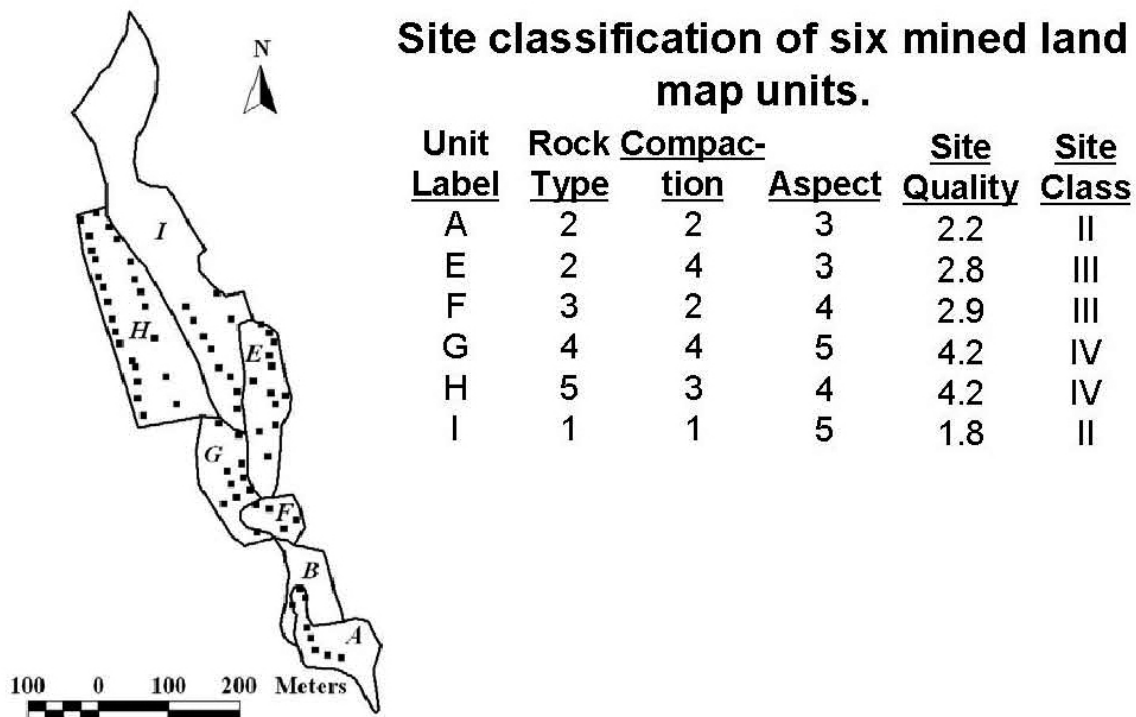


Figure 3-2-2. Soil/site quality classification and mapping of the Rapoca reclamation site in southwestern Virginia. The data points on the map indicate locations of selected white oaks that were used to test the classification criteria. Values are based on the site quality class scheme in Figure 3-2-1.

Soil samples were taken to a 40-cm depth within a 50-cm radius from the base of each tree. Samples were air dried and passed through a 2-mm sieve, and an array of physical and chemical properties were measured. Bulk density was determined using the excavation method (Blake 1965), and coarse fragment content was determined by sieving using a 2-mm screen. Soil particle size was determined using the hydrometer method (Bouyoucos 1936). Total soluble salts were measured in a 1:5 soil:water mix with an electrical conductivity meter (Bower and Wilcox 1965), and pH was determined in a 2:1 water-soil suspension using a pH meter. Nitrogen (N) mineralization potential was measured following anaerobic incubation (Bremner 1965a), and inorganic N was determined using a 1.0-M KCl extraction method (Bremner 1965b). Total N and carbon (C) were measured using a dry combustion C-N analyzer (Vario MAX Instruction Manual 2000). Exchangeable cations were extracted using the ammonium acetate method and analyzed using an ICP spectrophotometer (SpectroFlame Modula Tabletop ICP 1997, Spectroanalytical Instruments, Germany; Thomas 1982). Available nutrients were found using the Mehlich I test, and available phosphorus (P) was measured using the sodium bicarbonate method (Olsen and Sommers 1982).

Site factors including aspect, slope, and distance from native forest were also determined for each tree. Aspect was scaled by assigning numerical values starting at 1 for a northeast aspect and decreasing in value in either direction around the compass to 0 for a southwest aspect.

At the end of the growing season, the trees were excavated and soil adjacent to tree roots was used for the characterization of soil microbial properties (Table 3-2-1). Soil was kept at 4°C and was processed within 2.5 weeks of collection. For an estimate of microbial activity, dehydrogenase concentration was determined using a 2,3,5-triphenyltetrazolium chloride (TTC) indicator (Tabatabai 1982). Microbial biomass was measured by chloroform fumigation (Anderson and Domsch 1978, Gregorich et al. 1990, Jenkinson and Powlson 1976).

Leaves were harvested in mid-August from the upper portion of the crown, dried at 65°C for 7 days, and ground to pass through a 1-mm sieve. Nitrogen was determined using the C-N analyzer referenced above. Potassium (K), calcium (Ca), magnesium (Mg), manganese (Mn), and P were determined by dry-ashing the samples, extracting the nutrient elements using a 6N HCl solution, and analyzing the extracts with the ICP spectrophotometer referenced above (Jones and Steyn 1973).

Data Analysis. Tree height was regressed with site quality class to determine if the classification criteria represented factors influencing tree growth (SAS 2004). A multiple regression analysis was conducted to identify soil and site properties exerting effects on tree height. The independent variables were transformed based on established relationships with tree growth (Henderson et al. 1990, Kiniry et al. 1983, Fisher and Binkley 2000). Potassium and available N were transformed using the square root function, and microbial biomass and pH were transformed using a natural log function. An arc sine function was used to transform aspect and silt+clay percent in order to achieve a normal distribution (Little and Hills 1978). Backward Cp and R-squared selection were used to eliminate variables based on multi-collinearity and biological significance. Two experimental trees were eliminated based on differences in growth pattern. One tree had a double stem, while the other was much larger than any of the other trees. Two additional outliers were eliminated based on soil properties. Both had P levels over 10 times higher than any of the other samples. Therefore, statistical analyses were performed with $n = 68$.

Foliar samples were analyzed for nutrient sufficiency and balance using the Diagnosis and Recommendation Integrated System (DRIS) (Beaufils 1973). This analysis involves comparing the variance in foliar nutrient levels and nutrient ratios between the subpopulation of trees that are most successful (“reference population” = upper quartile) and the rest of the population. Ratios of the reference population were used as a standard by which imbalances in the remaining trees were assessed. A nutrient index for a given foliar nutrient level ‘A’ is developed using the following equation:

$$\text{Nutrient 'A' index} = \frac{[f(A/B) + f(A/C) + \dots + f(A/N)]}{z}$$

where

$$f(A/B) = ((A/B)/(a/b) - 1) \times (1000/CV) \text{ when } A/B \geq a/b,$$

or,

$$f(A/B) = (1-(a/b)/(A/B)) \times (1000/CV) \text{ when } A/B < a/b,$$

where A through N are the levels of foliar nutrients of interest, A/B is the nutrient ratio of the target population, a/b is the nutrient ratio of nutrient A to nutrient B for the reference subpopulation, z is the number of functions in the equation, and CV is the coefficient of variation of the reference population (Walworth and Sumner 1987). The closer the nutrient index is to

zero, the closer it is to the level of the reference population or the optimum. If it is negative, the nutrient is deficient, and if it is positive, the nutrient is in excess. The sum of the absolute values of the nutrient indices were added to obtain a nutrient balance index (NBI). The higher the number, the greater the nutrient imbalance and the poorer the tree growth. If the NBI is close to zero, foliar nutrient concentrations are close to optimal.

This analysis can be used to develop nutrient norms for a given plant species. Populations that are fertilized at various levels are compared, and the nutrient levels in plants that are most successful are used as norms. In this study, we used foliar nutrient levels to compare relative nutrient deficiencies and imbalances within a single population of young white oak trees. Nutrient means of the upper quartile of the population based on tree height were used as the standard against which foliar levels were compared. This analysis was done to find relative levels of nutrient imbalances among trees on the study site. It was simply used to distinguish between the nutrient levels of trees that were growing relatively well on certain sites and trees that were growing poorly.

Results and Discussion

Average values for total tree height and third-year growth for each mapping unit were plotted as a function of site quality. Results for total tree height and third-year growth were virtually the same; therefore, total tree height was used for all evaluations (Fig. 3-2-3). We hypothesized that tree growth would decrease as site quality decreased, where Class I is highest quality and Class V is lowest. The data show a trend, but the relationship between tree growth and designated site quality for the mapping units was not significant ($p > 0.1$). Measured tree growth showed that site quality was incorrectly assigned to several of the mapping units. This showed that the classification criteria used in the proposed model do not represent all the soil and site factors influencing tree growth, and that additional classification criteria are needed to adequately estimate site quality.

The spectrum of measured site and soil physical, chemical and biological properties hypothesized to influence tree growth is presented in Table 3-2-1. In nearly all cases, values ranged considerably among the sampled points. For example, slope steepness ranged from 0 to 55 and slope aspect covered the spectrum from northeast to southwest. Bulk density of the fine earth fraction ranged from 0.61 to 1.39 g cm⁻¹ and silt plus clay ranged from 17 to 64%. Chemical properties were also highly variable, with soil exchangeable nutrients ranging by an order of magnitude. Spoil pH ranged from 3.2 to 7.8 and EC ranged from 0.14 to 4.72 dS m⁻¹. Microbial activity was virtually absent on some sites and quite prevalent on others. As a result of this broad range in soil and site properties, tree height ranged from 15 to 125 cm.

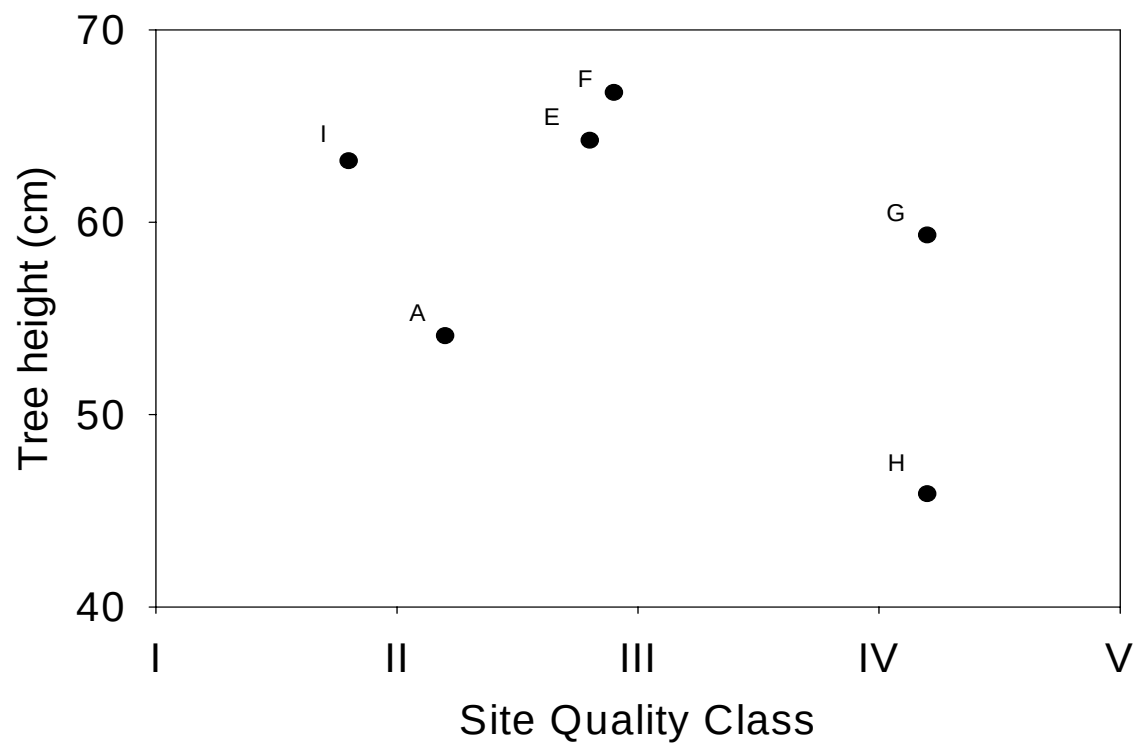


Figure 3-2-3. Mean tree height as a function of soil/site quality for mapping units based on a preliminary classification scheme proposed by Burger et al. (2002).

Table 3-2-1. Mean values and ranges for tree height and soil/site properties associated with each tree on the study site.

Variable	Mean	Min	Max	St Dev	CV	
Tree Height (cm)	56.4	15.2	125.0	22.5	40.0	
Site Properties						
Aspect †	0.27	0.0	1.0	0.29	107.44	
% Slope	24	0	55	16	69	
Dist from Native Forest (m)	69	6	162	41	59	
Physical Properties						
% Silt + Clay	45	17	64	9	20	
% rock fragments	35	21	51	6	16	
Bulk Density (fine earth) (g cm ³ ⁻¹)	0.96	0.61	1.39	0.18	18.27	
Chemical Properties						
EC (dS m ⁻¹)	0.92	0.14	4.72	1.03	112.29	
pH	5.2	3.2	7.8	1.5	27.8	
Anaerobic N (mg kg ⁻¹)	9.96	0.49	62.95	10.12	101.66	
KCl extractable N (mg kg ⁻¹)	5.20	1.59	21.55	3.52	67.78	
P (mg kg ⁻¹)	7.24	0.92	34.86	5.75	79.44	
Melich I Available Nutrients (mg kg ⁻¹)	K	33.5	8.7	72.3	13.2	39.5
	Ca	542.3	54.0	1426.4	368.3	67.9
	Mg	142.0	24.9	267.2	62.6	44.1
	Zn	2.2	0.8	5.9	1.1	50.6
	Mn	19.8	6.1	52.2	11.1	56.0
	Cu	1.1	0.4	1.8	0.3	25.9
	Fe	41.9	12.0	138.9	24.3	58.0
	B	0.09	0.05	0.20	0.04	47.67
Total N (%)	N	0.05	0.03	0.15	0.02	41.95
Total C (%)	C	1.08	0.30	4.39	0.76	70.53
Exchangeable cations (mg kg ⁻¹)	Ca	252.5	44.5	539.9	117.1	46.4
	Mg	100.5	27.9	147.0	23.2	23.1
	K	161.5	28.4	343.6	73.2	45.3
	Na	65.1	16.0	107.2	17.0	26.1
CEC (cmol+ kg ⁻¹)	3.01	0.99	5.65	1.11	37.02	
Biological Properties						
Dehydrogenase (mg kg ⁻¹)	11.42	0.25	80.83	15.47	135.46	
Microbial biomass (mg kg ⁻¹) (dissolved organic carbon)	38.1	3.3	201.8	35.8	93.9	

† aspect was calculated using a continuum where southwest = 0 and northeast = 1.

To better understand which soil and site factors were influencing white oak seedling growth, tree height was regressed against soil and site properties using data for all 68 trees. From greatest to least importance in the regression model, microbial biomass, pH, extractable soil K, total extractable inorganic N, % silt + clay, aspect, and extractable soil P were found to be the most important factors influencing white oak growth on this site (Table 3-2-2). Over 52% of the variation in tree height was described by the seven soil and site factors:

$$\text{Tree ht} = 33.10 + 9.19\ln(\text{bio}) - 29.01\ln\text{pH} + 6.61\sqrt{(\text{K})} + 10.05\sqrt{(\text{N})} - 61.15\arcsin(\text{siltclay}) + 11.33\arcsin(\text{as}) + 0.02(\text{P})^2$$

$$\text{R}^2=0.5205$$

Table 3-2-2. Variables and standardized coefficients for the regression model describing 52% of the variation in tree growth.

Variable	Standardized coefficient	p value
<i>Bio</i> = biomass of microbes (mg L ⁻¹)	0.40622	0.004
<i>pH</i> = pH (-log[H ⁺])	-0.40334	0.0068
<i>K</i> = potassium (mg kg ⁻¹)	0.38567	0.0019
<i>N</i> = total extracted nitrogen (mg kg ⁻¹)	0.35289	0.0048
<i>Siltclay</i> = % silt + clay	-0.26324	0.0345
<i>as</i> = aspect (0 to 1 scale SW to NE)	0.24696	0.0318
<i>P</i> = phosphorus (mg kg ⁻¹)	0.19187	0.0796

Mean values of the significant soil properties varied widely among mapping units (Table 3-2-3). These areas are easily distinguished by their vastly different physical, chemical and biological properties, suggesting that their map unit delineation was warranted. However, these properties also varied considerably within mapping units.

Table 3-2-3. Mean values for height and soil/site properties that were included in the regression model for each mapping unit.

Variable	Mapping Unit					
	F	E	I	G	A	H
Height (cm)	66.74	64.26	63.19	59.33	54.10	45.89
Microbial biomass (mg kg ⁻¹) (chloroform fumigation)	84.15	32.51	38.96	64.38	66.95	16.59
pH	6.57	4.64	4.75	6.22	6.19	4.96
Melich I Available K (mg kg ⁻¹)	33.22	36.04	39.47	30.17	45.71	29.09
Available N (mg kg ⁻¹)	11.19	4.02	4.99	8.04	6.47	3.43
% Silt + Clay	42.93	50.39	49.34	43.39	47.50	39.84
Aspect [†]	0.13	0.73	0.19	0.00	0.00	0.17
P (mg kg ⁻¹)	7.12	6.77	9.31	6.52	8.88	6.63

[†] Aspect was calculated using a continuum where southwest = 0 and northeast = 1.

Microbial biomass was represented with a mean of 38.1 mg kg⁻¹ of microbial organic C; it was skewed toward lower levels, with many sites having virtually no microbes, and only a few sites having levels above 100 mg kg⁻¹ (Fig. 3-2-4). Microbial biomass was positively correlated with tree growth. Since correlation does not necessarily imply causation, it is unknown whether microbial populations have a significant effect on tree growth or whether microbial biomass responded to the site factors that influenced tree growth. The fact that microbial biomass was independent of other regressors in the model suggests that it is a causative factor. Symbiotic mycorrhizal colonization of the tree root systems would be an example of causation. In any case, microbial activity is a good indicator of soil quality (Miller 1998), as microbes play important roles in decomposition of organic matter, N mineralization and fixation, and symbiosis. Hutson (1980) found that low population densities of organisms on industrial reclamation sites led to significantly less degradation of oak leaves than on control sites. Microbes are the main

mechanism for the release of plant-available nutrients from organic matter (Brady and Weil 2002). In his discussion of techniques for reclaiming with native hardwoods, Miller (1998) stated that the development of a healthy and diverse soil microbial population through replacement of topsoil is essential to the establishment of native trees.

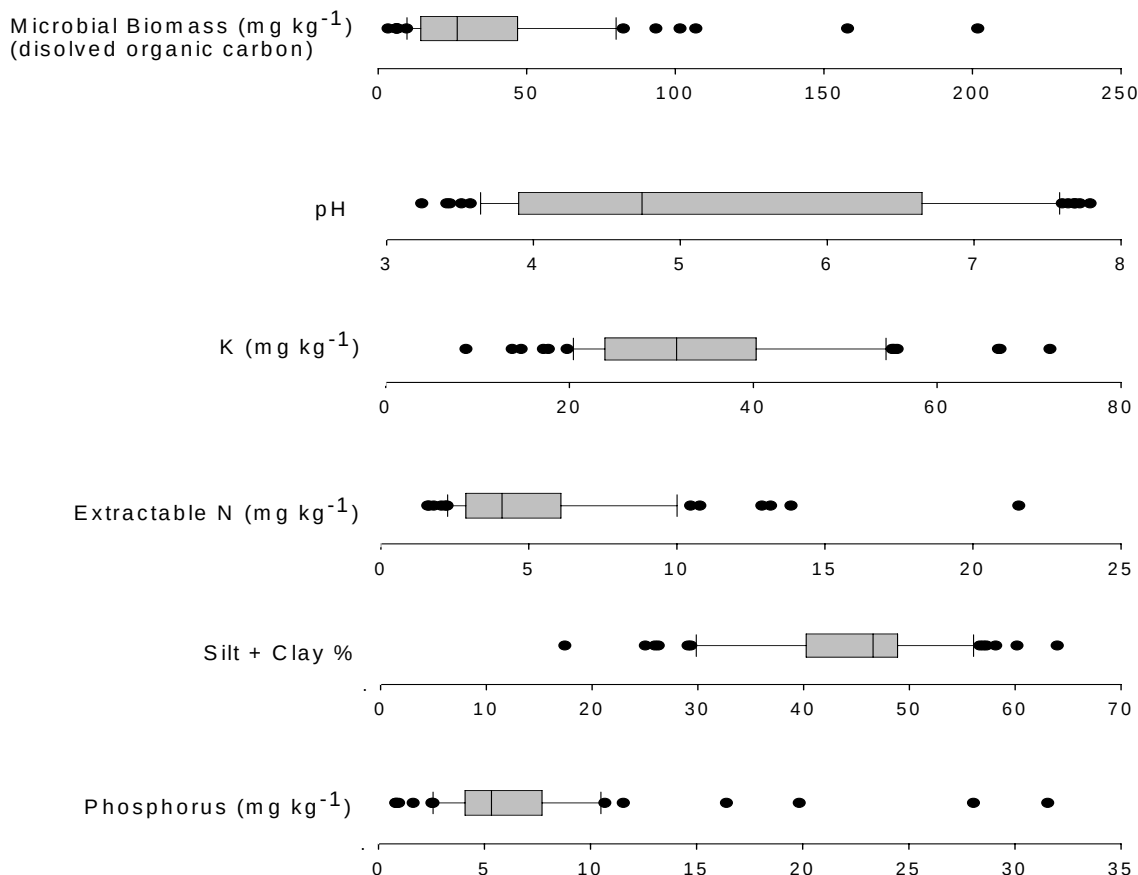


Figure 3-2-4. Box plots for variables in the regression model, listed from most to least important to tree growth.

Mine soil pH ranged from 3.2 to 7.8 with a mean of 5.2 (Fig. 3-2-4). It had a significant negative correlation with tree growth and was the second most important variable in the regression model. Mine soil pH is related to many other chemical and biological properties. It is closely related to spoil type and is a master variable determining nutrient availability (Brady and Weil 2002).

The average level of soil N measured across the study area was 5.20 mg kg⁻¹ (Fig. 3-2-4). This level is considered deficient (Fisher and Binkley 2000), suggesting that the higher levels found on some sites would have a positive impact on tree growth. Levels of K and P ranged widely, with P skewed toward smaller amounts while K was more evenly distributed. Soil P, K, and N were positively correlated with tree growth and were found to be statistically significant components of the regression model. Tree growth was a function of the square root of the level of N and K and the level of P squared, which are typical response relationships. Nutrients that

were collinear with K, including Ca and Mg, were removed from the regression analysis; therefore, K may be seen as representing a group of cations important to tree growth.

Soil types ranged from loamy sands to sandy loams with 17 to 65% silt + clay (Fig. 3-2-4). Percent silt + clay was negatively correlated with tree growth. This negative correlation suggests that oak trees grow better in sandier mine soils. This effect may be caused by low hydraulic conductivity and poor drainage in the more finely-textured mine soils, or it may be because silt + clay content correlates with chemical factors that influence tree growth. Torbert et al. (1990) also found a strong positive correlation between percent sand and tree growth.

Aspect ranged from northeast to southwest. Tree height was positively correlated to aspect increasing from southwest to northeast. Northeast aspects are more mesic and cooler during the growing season, making them more conducive to tree growth.

Because soil fertility factors played a dominant role in the model of tree growth as a function of soil and site properties, (Eq. 2), foliar nutrient concentrations were analyzed to better understand the role of soil fertility on tree growth.

The nutrient levels of the reference population were much closer to levels found in foliage from white oaks growing in the native Appalachian hardwood forest. Auchmoody and Hammock (1975) found levels of 25.6 mg kg⁻¹ for N, 1.5 mg kg⁻¹ for P, 7.3 mg kg⁻¹ for K, 4.7 mg kg⁻¹ for Ca, 0.9 mg kg⁻¹ for Mg, and 1.0 mg kg⁻¹ for Mn (Table 3-2-4, col. 2). These trees were located on a mixed oak stand in the Fernow Experimental Forest in Parsons, West Virginia. The upland oak site index was 66 and the stand was 60 years old. Foliage samples were taken from the upper part of the crown in late August. A comparison of these nutrient levels with those occurring on our study site shows that foliar N levels are insufficient on the mine site, while P and K are sufficient and Ca, and Mg are in excess compared to these native forest levels.

DRIS nutrient analyses showed that N, K, and Ca were deficient or in excess and that ratios of N/K, K/Ca, and K/Mg were out of balance (Table 3-2-4). This suggests that levels of N, K, P, Mg, and Ca are all influencing the growth of these trees on these sites. The nutrient indices suggested that trees on all mapping units were deficient in N and P and had an excess or sufficiency in Ca and Mg (Table 3-2-5). K was either sufficient or slightly deficient, depending on the mapping unit.

Table 3-2-4. DRIS foliar nutrient ratios for good and poor populations of three-year-old white oaks on a mine site in Virginia.

Nutrient	Native unfertilized white oak nutrient levels [†]	High height growth [‡]		Low height growth		F-test for variance	
		Mean	CV	Mean	CV	ratio	p value
N	25.60	22.75	12.62	17.99	28.82	3.26	0.0104*
P	1.50	1.74	21.97	1.30	40.37	1.88	0.1575
K	7.30	7.48	19.52	8.03	30.03	2.73	0.0272*
Ca	4.70	8.73	29.87	10.93	46.77	3.84	0.0039*
Mg	0.90	1.75	43.61	2.56	42.75	2.07	0.1045
Mn	1.00	2.09	67.47	1.41	112.0	1.25	0.6329
N/P	17.07	13.54	19.78	15.02	35.96	4.07	0.0028*
N/K	3.51	3.12	17.10	2.53	45.93	4.74	0.0010*
N/Ca	5.45	2.81	27.97	2.02	53.66	1.91	0.1491
N/Mg	28.44	15.53	43.70	8.77	60.79	0.62	0.1994
N/Mn	25.60	35.43	166.7	50.04	121.3	1.05	0.9450
P/K	0.21	0.24	31.82	0.17	46.24	1.08	0.9226
P/Ca	0.32	0.21	30.27	0.15	57.43	1.76	0.2063
P/Mg	1.67	1.20	51.00	0.65	69.86	0.54	0.1017
P/Mn	1.50	2.37	160.6	3.22	112.0	1.12	0.7340
K/Ca	1.55	0.95	39.14	0.92	65.36	2.62	0.0332*
K/Mg	8.11	5.42	60.08	3.79	56.48	0.43	0.0245*
K/Mn	7.30	3.59	0.29	5.72	0.27	1.13	0.7203
Ca/Mg	5.22	5.55	32.37	4.56	35.81	0.83	0.5907
Ca/Mn	4.70	4.19	0.44	7.78	0.42	1.04	0.9934
Mg/Mn	0.90	0.84	0.65	1.82	0.38	0.70	0.3268

[†] From Auchmoody and Hammock, 1975.

[‡] Upper quartile of the population.

* Significance at a 0.1 level.

Table 3-2-5. Mean nutrient indices for three-year-old white oaks located across six mapping units.

Map Unit	N Index	P Index	K Index	Ca Index	Mg Index	NBI
H	-37.95	-30.66	17.59	18.86	32.16	137.22
A	-4.08	-13.04	-9.68	0.47	26.34	53.61
E	-3.97	-3.22	-4.18	5.68	5.69	22.74
I	-10.51	-14.17	6.26	10.49	7.93	49.36
G	-4.36	-16.31	-2.09	7.87	14.88	45.51
F	-6.40	-23.05	-7.46	10.62	26.29	73.82

When these nutrient indices were added (absolute values) using the DRIS technique to create a nutrient balance index (NBI), the values show that there is a relationship between the extent of imbalance and the amount of tree growth ($p = 0.0001$) (Fig. 3-2-5), despite the fact that many other factors besides fertility are also affecting tree height. The dashed line represents a nutrient imbalance above which further height growth is unlikely (Walworth and Sumner 1987). Tree height is severely limited when the NBI is greater than 200. Mapping unit H had the lowest tree height growth levels, the lowest predicted site quality, and the highest nutrient imbalances. This further supports the importance of nutrient levels in the designation of spoil quality and the ability of trees to grow in certain spoil materials. These DRIS results strongly support the inclusion of an indicator of soil available N, P, and K in the classification model.

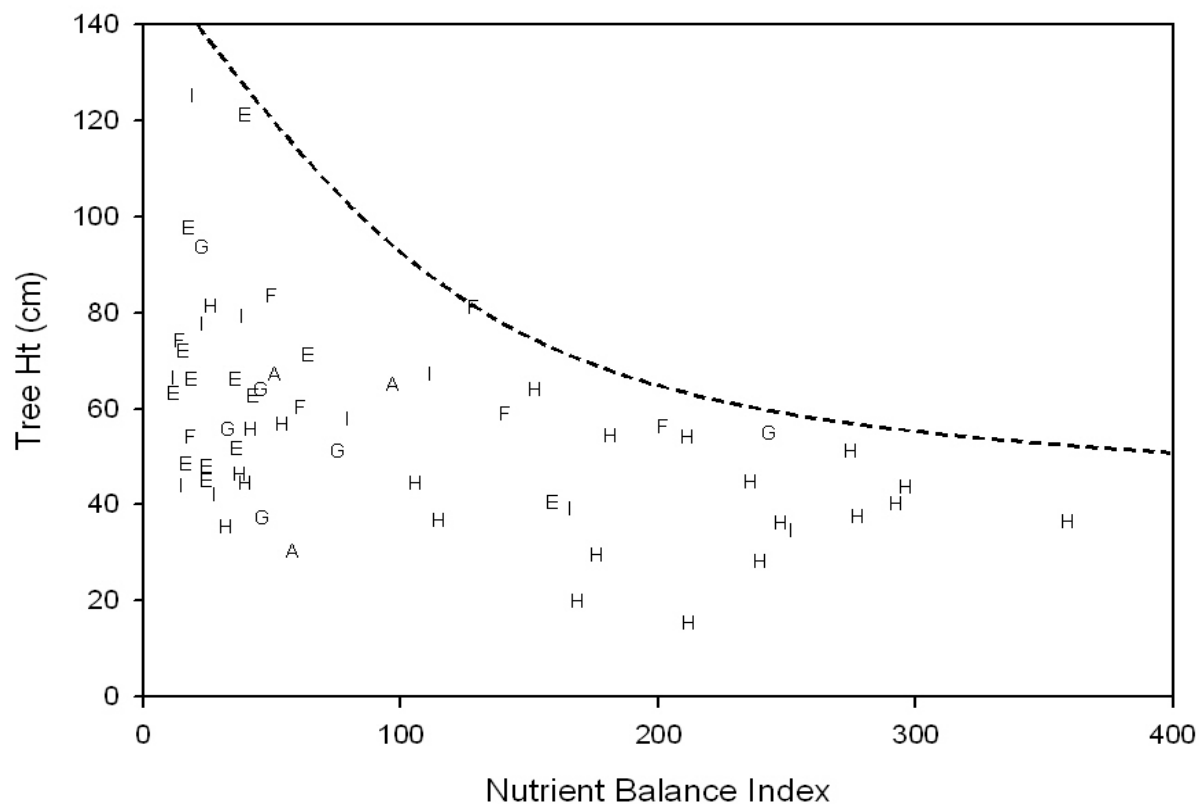


Figure 3-2-5. Tree height as a function of nutrient balance for 65 three-year-old white oak seedlings across six mapping units. The dashed boundary line represents a tree height limit for a given NBI.

The regression model and nutrient analysis show that the factors in the conceptual model for mine soil quality (soil strength, air/water balance, soil temperature, and nutrient levels) were not all limiting growth on this site. There were no variables in the final regression model indicating that density was a problem, suggesting that these mine soils were uncompacted. Air/water balance was represented in the regression model through the variables silt + clay and aspect, suggesting that water availability and aeration influenced white oak growth on this site. Soil temperature was represented in the regression by aspect. Nutrient availability was also an influential soil attribute on this site. It appeared in the regression model through soil variables

characterizing N, P, and K concentrations. The importance of available nutrients was further supported by the foliar DRIS analysis, where NBI levels were inversely proportional to tree growth.

Thus, three factors of the factors represented by the regression model – air/water balance, soil temperature, and nutrient levels – were important to white oak growth on this site. Air/water balance and soil temperature effects were represented by the mapping criteria, but nutrient levels were not. Although rock type influences nutrient availability to some degree, it alone did not represent the variability of nutrient levels across sites. An added criterion that better represents nutrient availability is thus needed in the mine soil classification model.

Nutrient levels, although established as important criteria of soil quality, are not easily measured in the field and thus may not be useful additions to a field mapping tool. However, the essential soil nutrients influencing white oak growth on this site, according to both the regression model and the DRIS analysis, were correlated with pH (Table 3-2-6).

Table 3-2-6. Correlation coefficients of pH with soil nutrient levels and other soil properties on the study site.

	Correlation Coefficient	Significance
Total Extracted N (mg kg ⁻¹)	0.42067	0.0002
K (mg kg ⁻¹)	0.38222	0.0009
P (mg kg ⁻¹)	-0.23450	0.0507
Ca (mg kg ⁻¹)	0.77269	<0.0001
Mg (mg kg ⁻¹)	0.86313	<0.0001
% Silt + Clay	-0.24561	0.0376
Microbial Biomass (mg kg ⁻¹)	0.54324	<0.0001

The significant correlation of pH with nutrients and tree growth suggests that an intermediate pH may be optimal. Soil reaction has been used in the past to assess mine soil quality for agronomic crops, but it is often modeled as a positive linear relationship with productivity for all situations, where a higher pH indicates a better soil. Although pH levels up to 6.5 can lead to an increase in P availability and CEC, pH above this level can decrease the availability of some micronutrients (Brady and Weil 2002) and shift microbial composition, especially mycorrhizal relationships on which native trees depend (Killham 1994). High soil pH increases herbaceous competition. At high pH, grasses have a distinct advantage and often out-compete trees (Burger and Zipper 2002). Furthermore, pH was not well correlated to rock type on this site, varying widely within each mapping unit. For these reasons, we recommend that pH be added as an additional criterion with equal weighting for soil classification and mapping as shown in the diagram in Figure 3-2-6. We expect that incorporation of pH will utilize a transformation that assigns maximum values to pH at mildly acidic levels that favor P availability but without stimulating excess herbaceous vegetation growth. Further testing of this modified field classification model on additional sites is needed to validate its usefulness and weight the relative importance of the four criteria for major reclamation tree species.

	5	4	3	2	1
Rock Type (WF _{wo} = 0.25)	unweathered shale 10:90	30-70	mixture 50:50	70:30	weathered 90:10 sandstone
	5	4	3	2	1
Compaction (WF _{wo} = 0.25)	high penetration 10 cm	20 cm	medium penetration 30 cm	40 cm	low penetration 50 cm
	5	4	3	2	1
Aspect (WF _{wo} = 0.25)	SW 180-270°	270-315° 135-180°	Flat	315-360° 90-135°	0-90° NE
	5	4	3	2	1
Soil pH (WF _{wo} = 0.25)	3.5 – 4.4	7.5 – 8.4	6.5 – 7.4	4.5 – 5.4	5.5 – 6.4
	V	IV	III	II	I
Site Quality Class $[SQC = \sum_{i=1}^4 (\text{site factor rating} \times WF)]$					

Figure 3-2-6. Modified site quality classification model showing the addition of a pH criterion to help estimate forest site quality class (SQC) for white oak seedlings. An equal weighting factor (WF) of 0.25 was assigned to all four classification criteria pending further testing.

Conclusions

This study tested a mine soil classification system that included rock type, compaction, and aspect as field mapping criteria (Burger et al. 2002). The study showed that additional criteria representing mine soil chemistry and fertility are needed for adequate characterization of mine site quality for white oak trees.

Our regression model of tree height as a function of site and soil properties showed that sandy loam soils with a northeast aspect, high nutrient levels, and high microbial populations were the most conducive to growth of young white oaks. Of these properties, soil particle size and aspect were represented in the existing classification model and accounted for the range in water/air balance (Burger et al. 2002). Nutrient availability was also a growth-determining factor, but it was not a component in the original classification model. Nutrient availability was well correlated with spoil pH, pH was correlated with tree growth, and pH is an easily-measured field property that could be included as a field classification criterion. Therefore, based on white oak response to mine soil properties, we recommend that pH be included as a classification criterion for mine soil mapping along with rock type, soil density, and aspect. A revised classification model will need further testing with an independent data set.

With species-specific soil quality maps, native tree species can be selected for planting based on the given quality of a reclaimed mined site, leading to increased survival and growth, more timely bond release for mine operators, and improved success restoring healthy native hardwood stands.

Task 3, Sub-study 3

Experimental

The study site was a white pine plantation located on a surface mine in Wise County, Virginia, reclaimed prior to the passage of the Surface Mining Control and Reclamation Act of 1977. Following surface mining of the coal, the overburden rock was simply pushed back across the site, creating a bench and highwall profile. The resulting spoil at this site was a mixture of sandstone, siltstone, and coal-derived materials. The stand was planted in 1978, and in 1996 a thinning study was installed. In 1996, at age 17, the stand contained 1438 stems ha^{-1} with 30.1 $\text{m}^2 \text{ha}^{-1}$ of basal area. Three paired blocks of 0.02-ha plots were established in the stand prior to thinning. One plot in each block was randomly selected for thinning, and the basal area was reduced to 20.7 $\text{m}^2 \text{ha}^{-1}$, leaving a final stand density of 652 stems ha^{-1} . The second plot in each pair was left as a control and was not thinned. All plots were measured in 1996 for total height and diameter at breast height for all living white pines. Five randomly selected dominant or co-dominant trees from each plot were measured annually to evaluate the change in diameter increment due to thinning over time. Height and diameter of all trees in the plots were re-measured in 2004, nine years after the thinning, when the stand was 26 years old.

Site index was calculated based on the average height of trees in the upper quartile of total height to approximate dominant and co-dominant trees using site index equations for white pine in the southern Appalachians (Beck 1971). Cubic foot volumes to a 10 cm top d.i.b. were calculated using volume equations for white pine in the southern Appalachians (Vimmerstedt 1962). Board foot volumes to a 15cm top d.i.b. were calculated using equations for white pine plantations in southeastern Ohio (Dale et al. 1989). Minimum diameter for sawtimber was set at 30 cm. For pulpwood, cubic meter volumes were converted to cubic foot volumes and then to tons of pulpwood using a conversion factor of 35.9 $\text{ft}^3 \text{ton}^{-1}$. Stumpage prices for sawtimber and pulpwood (Timber Mart-South 2004) were applied to stand volume estimates to obtain stand value estimates for thinned and unthinned treatments. Stand parameters measured at age 26 were projected to age 30 using a stand table projection (Avery and Burkhart 2002). Total tree height at age 30 was predicted from site index equations (Beck 1971) based on the site index of each tree at age 26.

Data from the 2005 inventory were analyzed for differences in dbh, basal area per hectare, trees per hectare, volume per hectare, volume per tree, proportion of volume in sawtimber, and value per hectare between treatments using a random complete block design with three blocks and two treatments per block. Analysis of variance was used to detect statistically significant differences between treatments. Proportion data were transformed using arcsine transformation prior to analysis of variance. The annual diameter measurements were analyzed using a repeated measures mixed model procedure to test the statistical significance of change in diameter increment with respect to thinning treatment over time. Transformation of the response variable using the natural log function was used to satisfy model assumptions. SAS version 8.2 (SAS Institute Inc., Cary, NC 2001) was used for all statistical analyses and significance was set at $P < 0.05$ for all comparisons.

Results and Discussion

Site index for the stand averaged 32.0 m at base age 50 years using equations from Beck (1971) for white pine in the southern Appalachians. This is well above the site index noted by

Doolittle (1962), who found average site index for white pine in the southern Appalachians to be 24.4 m at base age 50. Dale and coworkers (1989) reported average site index of white pine in southeastern Ohio to be 23.5. The response to thinning from age 17 to age 26 is shown in Table 3-3-1. As expected, total height of the thinned treatment was greater than that of the unthinned treatment for both ages due to the removal of intermediate and suppressed trees from the plots treated with low thinning. Thinning increased dbh by nearly 4.5cm over the nine years since treatment compared to the unthinned treatment (27.9 cm vs. 23.4 cm for these treatments, respectively). The annual diameter increment calculated from the repeated measures data for the thinned treatment was 0.84 cm yr⁻¹, while that for the unthinned treatment was 0.58 cm yr⁻¹, and this difference was significant ($P < 0.0001$) (Figure 3-3-1). Basal area was not significantly different between treatments. Gillespie and Hocker (1986), in a study of white pine thinning response in New England, found that stand basal area was not affected by thinning, but mean diameter increment was significantly greater in the thinned plots. Both treatments have accrued a large amount of basal area (15.6 m² ha⁻¹ and 12.9 m² ha⁻¹ for thinned and unthinned, respectively). Comparing the stand density prior to thinning (1438 stems ha⁻¹) with the stand density in the unthinned treatment, it can be seen that substantial mortality has taken place in the unthinned treatment, as there remains only 63% of the original number of trees in the unthinned plots. Low thinning has been shown to decrease competition-induced mortality for white pine in the southern Appalachians (Della-Bianca 1981). The volume per acre in the thinned treatment was not significantly different compared to the unthinned treatment. Additionally, the volume of the thinned plots is standing volume and does not account for the 94.2 m³ ha⁻¹ removed during the thinning. Individual tree volume was significantly different between treatments at age 26 ($P = 0.0327$), which is reasonable given the diameter growth response observed.

Table 3-3-1. Thinning effects at age 26, nine years after thinning, and projected thinning effects at age 30 for a white pine stand growing on a reclaimed surface mine in southwestern Virginia.

Treatment	DBH (cm)	Total Height (m)	Basal Area (m ² ha ⁻¹)	Trees per Hectare	Stand Volume (m ³ ha ⁻¹)	Volume per Tree (m ³)	Volume in Sawtimber (%)	Value per Hectare (\$)
Age 26:								
Thinned	27.9	19.3	36.3	566	289.6	0.52	62	5641
Unthinned	23.4	17.0	42.9	899	312.7	0.35	55	5481
Pr > F	0.018	0.017	0.111	0.044	0.360	0.033	0.514	0.593
Age 30:								
Thinned	33.0	22.1	49.8	566	465.6	0.84	92	11265
Unthinned	26.2	19.7	53.0	899	456.9	0.51	66	8807
Pr > F	0.007	0.018	0.415	0.044	0.796	0.015	0.015	0.008

At age 26, low thinning had not created a significant difference in the proportion of stand volume in the sawtimber size classes compared to the unthinned treatment, but with the continued increased diameter growth rates (Fig. 3-3-1) this shift would be expected in the near future. Stand table projection was used to predict the stand parameters at age 30. This projection indicated that there would be nearly a three-inch difference in dbh between the thinned and unthinned treatments, which was statistically significant ($P = 0.0066$) (Table 3-3-1). Due to the accelerated diameter growth, stand basal area would be very similar for thinned (49.8m²) and unthinned (53.0m²) treatments.

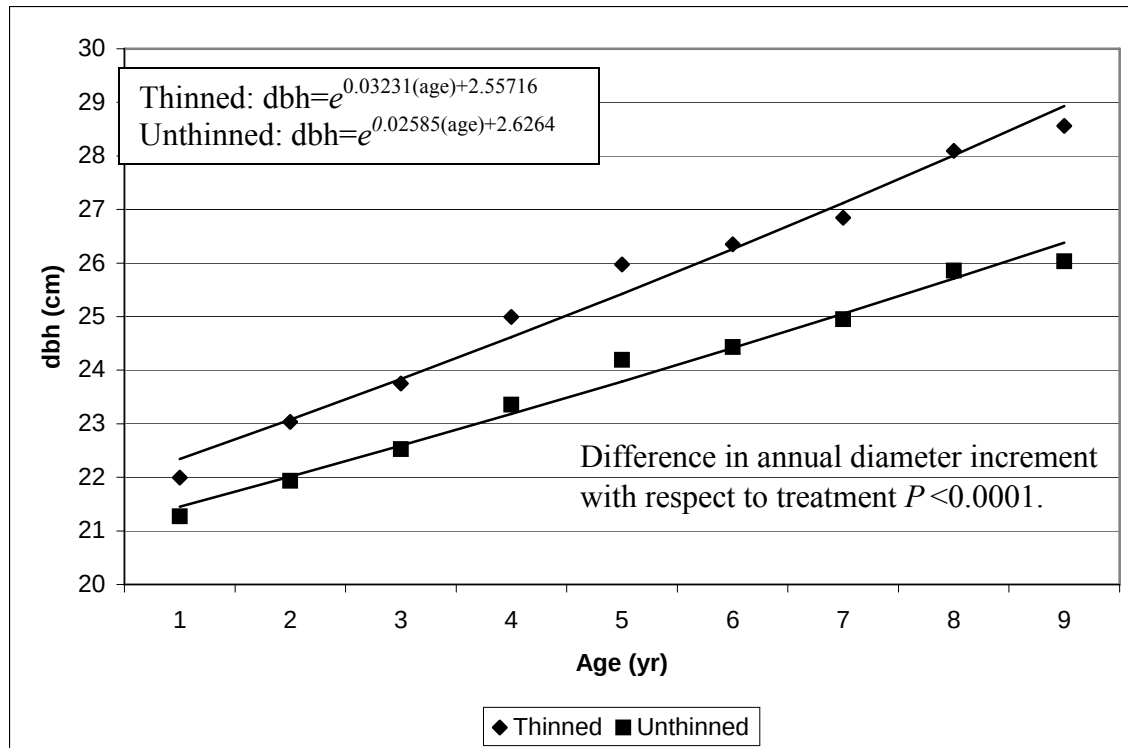


Figure 3-3-1. Diameter response to thinning of a 26-year-old white pine stand growing on a reclaimed surface mine site in southwestern Virginia.

At age 30, standing volume in thinned plots was estimated to be $465.6 \text{ m}^3 \text{ ha}^{-1}$, which would surpass the volume of the $456.9 \text{ m}^3 \text{ ha}^{-1}$ in the unthinned treatment. Volume per tree was estimated to be nearly $0.34 \text{ m}^3 \text{ tree}^{-1}$ more in the thinned treatment than the unthinned treatment at age 30, compared to an approximate $0.17 \text{ m}^3 \text{ tree}^{-1}$ difference between the same treatments, respectively, at age 26. It is estimated that at current diameter growth rates, 92% of the volume in the thinned treatment would be sawtimber compared to 66% for the unthinned treatment, and this difference would be statistically significant ($P = 0.0154$). Results of a white pine thinning study in the southern Appalachians found that at both 80 and 100 years, thinning had shifted the diameter distributions to larger size classes, but failed to increase cumulative yield compared to an unthinned control (Della-Bianca, 1981; McNab and Ritter, 2000). McNab and Ritter (2000) did note that site quality, as measured by site index, was higher in the unthinned control, indicating that if site qualities were equal, it might be possible for thinned plots to produce more cumulative yield. Due to higher stumpage values for sawtimber, the higher proportion of sawtimber in the thinned treatment would translate into a significantly higher value per acre for the thinned treatment ($P = 0.0079$), whereas at age 26, both treatments had similar standing volume and proportion of volume in sawtimber, and consequently there was no significant difference in value per acre at age 26. The magnitude of the shift into the sawtimber class for both treatments at ages 26 and 30 can be seen in Figure 3-3-2. The result of this shift is an approximate 200% increase in value for the thinned treatment from age 26 to age 30 (\$5641 and \$11265 ha^{-1} for ages 26 and 30, respectively) and an approximate 160% increase in value for the unthinned treatment (\$5481 and \$8807 ha^{-1} for ages 26 and 30, respectively).

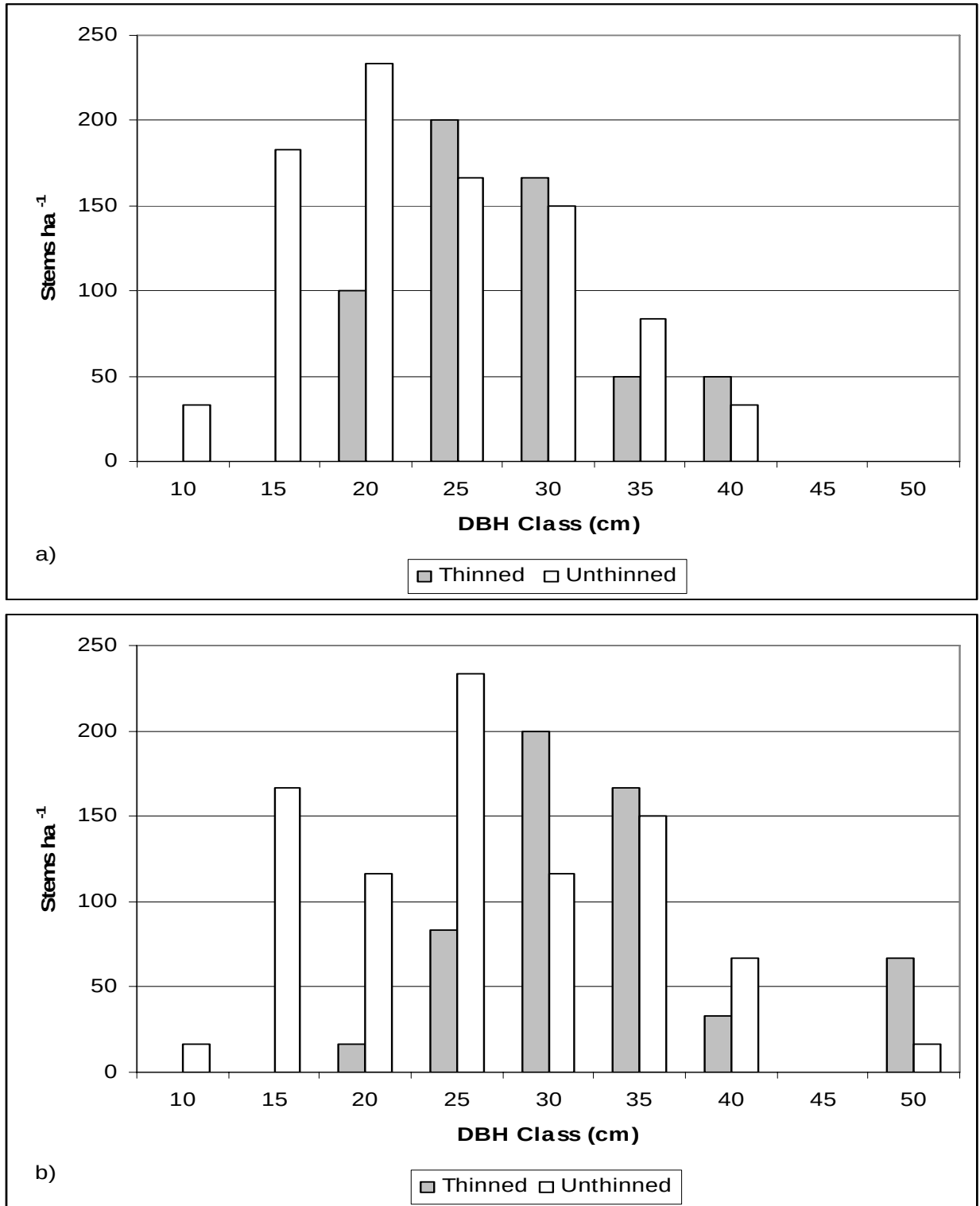


Figure 3-3-2. Diameter distributions for (a) stand at age 26 and (b) stand projected to age 30 for a 26-year-old white pine stand growing on a reclaimed surface mine in southwestern Virginia.

To examine the economic feasibility of establishing stands with this level of productivity on surface mined lands, it is important to understand that several factors have been found to limit tree productivity on post-SMCRA reclaimed surface mined land. The two major limitations are soil compaction and competing vegetation (Ashby, 1991) that result from SMCRA's requirement to return the land to approximate original contour and to stabilize the reclaimed landscape from erosion. Burger and Zipper (2002) have outlined procedures for restoring forests on surface mined lands. Part of their prescription includes the establishment of a tree-compatible ground cover, which is intended to minimize the need for competition control treatments when forestry is chosen as the post-mining land use; however, these treatments may still be required if other more aggressive herbaceous species become established on the site. As such, using the stand value information from this study, net present values (NPV) and internal rates of return (IRR) were calculated for four different management scenarios that are likely to face landowners who wish to establish forests on post-SMCRA mined lands. The NPV allows for comparison of the different scenarios while accounting for the opportunity costs associated with each investment scenario. Important assumptions include having a tree-compatible ground cover established by the mining company, having appropriate spoil materials for tree growth (Torbert and Burger, 2000), and having these materials returned to the surface in an uncompacted state. It was also assumed that the harvested volume resulting from thinning would cover the cost of the thinning operation at no net benefit or cost (harvestable volume would have generated \$667ha⁻¹ based on pulpwood prices and the cubic foot volume removed during thinning). A 6% discount rate was used in calculating NPV. The scenarios evaluated differ only with respect to costs incurred over the rotation and include:

- Scenario 1. Establishment costs only of \$618ha⁻¹ (Burger and Zipper, 2002)
- Scenario 2. \$618ha⁻¹ establishment cost and \$173ha⁻¹ herbicide cost (based on author estimates) in first year
- Scenario 3. \$618ha⁻¹ establishment cost and \$173ha⁻¹ herbicide cost in years 1 and 2
- Scenario 4. \$618ha⁻¹ establishment cost and \$173ha⁻¹ herbicide cost in years 1, 2, and 3
- Scenario 5. All establishment and herbicide costs borne by mining company up to year 5 to obtain bond release.

Cash flows for each scenario are depicted in Figures 3-3-3 and 3-3-4. The results of this simulation show that at age 26, the IRR's and NPV's are virtually the same between thinned and unthinned treatments for Scenarios 1 through 4 and ranged from approximately 9% when management inputs include only establishment costs to approximately 6.5% using the most intensive scenario (Table 3-3-2). Using projected values at age 30, the IRR's for Scenarios 1 through 4 differed by approximately 1% between the treatments, with the IRR of the thinned treatment being higher. For the thinned treatment, IRR's range from 10.2% to 8.1% from the least intensive to the most intensive scenarios respectively. For the unthinned treatment the range is 9.3% to 7.2% respectively. When commercial forestry is specified as the post-mining land use, mining companies are required by law to establish a minimum stocking of crop trees per hectare within a fixed time period to obtain bond release. In Virginia, 988 crop trees ha⁻¹ are required (Burger and Zipper, 2002). It is important to understand that scenario 5 is not typical for forestry business enterprises and represents a situation in which all harvest revenues are purely profit to the landowner as there are no establishment costs to be considered. This resulted in NPV values that were approximately \$600 ha⁻¹ greater under scenario 5 when compared to scenario 1, with this difference increasing as additional herbicide costs are incurred. Calculation of IRR for scenario 5 was not possible given that revenues were generated, but no cost were incurred, meaning that regardless of the interest rate, NPV could not be set equal to zero.

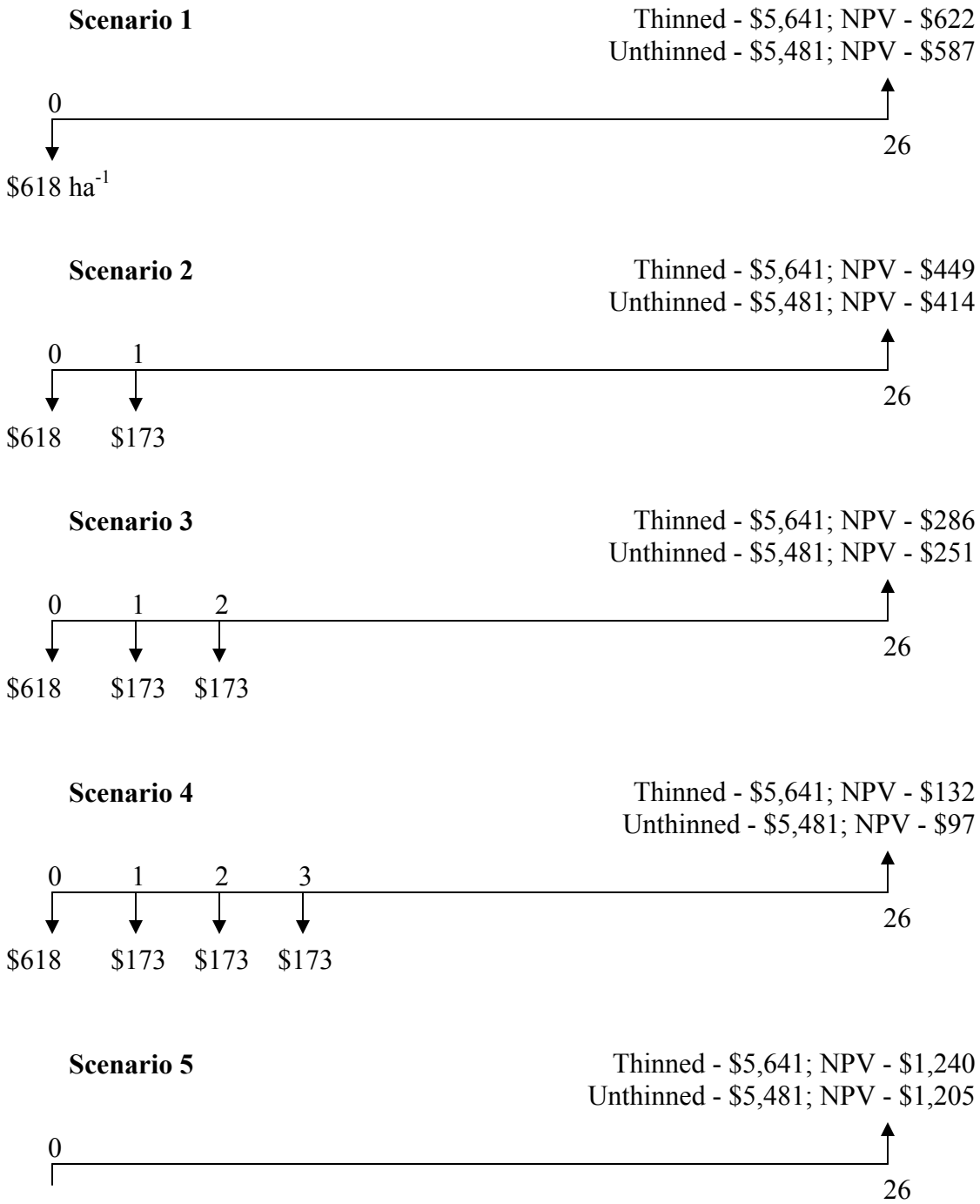


Figure 3-3-3. Cash flow diagrams for thinned and unthinned treatments under Scenarios 1-5 at age 26 for a white pine stand growing on a reclaimed surface mine in southwestern Virginia.

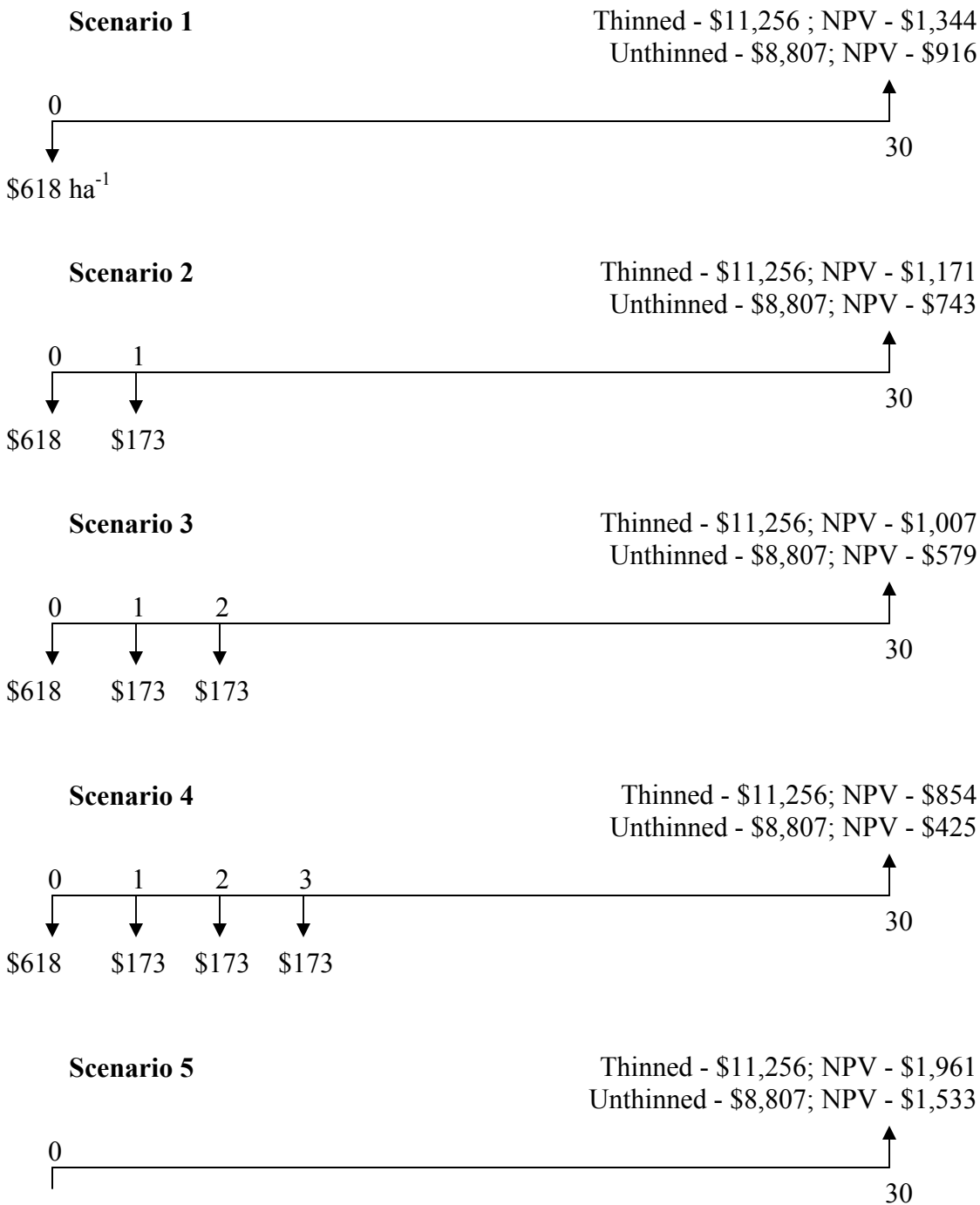


Figure 3-3-4. Cash flow diagrams for thinned and unthinned treatments under Scenarios 1-5 at age 30 for a white pine stand growing on a reclaimed surface mine in southwestern Virginia.

Table 3-3-2. Net present value (\$ ha⁻¹) at 6% interest and internal rate of return by thinning treatment and stand age for a 26-year-old white pine stand growing on a reclaimed surface mine in southwestern Virginia.

Age and Treatment	Management Scenario*									
	1		2		3		4		5	
	NPV (\$)	IRR (%)	NPV (\$)	IRR (%)	NPV (\$)	IRR (%)	NPV (\$)	IRR (%)	NPV (\$)	IRR (%)
Age 26:										
Thinned	622	8.9	449	7.9	286	7.1	132	6.5	1240	---
Unthinned	587	8.8	414	7.7	251	7.0	97	6.3	1205	---
Age 30:										
Thinned	1344	10.2	1171	9.3	1007	8.6	854	8.1	1961	---
Unthinned	916	9.3	743	8.4	579	7.7	425	7.2	1533	---

*Scenario 1: \$618ha⁻¹ establishment costs only

Scenario 2: \$618ha⁻¹ establishment cost and \$173ha⁻¹ herbicide cost in first year

Scenario 3: \$618ha⁻¹ establishment cost and \$173ha⁻¹ herbicide cost in years 1 and 2

Scenario 4: \$618ha⁻¹ establishment cost and \$173ha⁻¹ herbicide cost in years 1, 2, and 3

Scenario 5: All costs to age 5 paid by mining company

From this simulation, it can be seen that if sawtimber production is the management objective, and the desired rotation age is around 30 years, thinning near mid-rotation is a better economic decision than leaving the stand to grow in an unthinned state, especially considering that the average diameter of unthinned trees is only projected to be 26.2cm at age 30 and only 66% of the stand volume is in sawtimber.

Conclusions

The results of this investigation reveal that, at a site index of 32.3m, this stand is more productive than established averages for white pine in the southeastern United States. Additionally, volume growth rates of 11.1m³ ha⁻¹ yr⁻¹ in the thinned plots compare favorably with productive stands of loblolly pine found in the southeastern U.S., thus demonstrating the potential of reclaimed surface mines to support productive forests. Thinning the stand at age 17 rapidly increased the diameter growth of the residual trees. Volumes and values for the stand were no different at age 26; however, at each treatment's respective growth rates based on a stand table projection, stand values were significantly higher for the thinned treatment by age 30 due to a shift in the diameter distribution of this treatment toward the sawtimber size classes. These trends in terms of thinning response are similar to trends found in white pine stands growing on native soils, and as such, it appears that white pine growing on reclaimed surface mines can be managed similarly to plantations on native soils. Economic analysis of stand value information revealed that stands growing at this level of productivity on reclaimed mined lands should provide landowners with favorable returns on their investment even when establishment and weed control costs are borne by the landowner. When establishment costs for stands at this level of productivity are borne by the mining companies as part of the reclamation process as required by the SMCRA, the before-tax NPV the landowner could realize is \$1,961 ha⁻¹.

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TASK 4: Conduct economic analyses of reforestation and forest management activities for carbon sequestration and a variety of forest products and services.

Executive Summary

We examined the financial viability of reforesting reclaimed surface mined lands in the Appalachian coal-mining region, addressed the economic implications of a shift in reforestation burden from the landowner to the mine operator or government, and explored the opportunities for forest carbon payments on these lands. Our analysis is conducted by comparing land expectation value (LEV) for a range of land use conversion scenarios in which the landowner assumes the costs of site preparation and tree planting, versus scenarios in which the coal company assumes the costs. The comparisons are then used to determine the level of carbon payment required to make forest land-use a financially viable option in circumstances where it is not already viable.

Grassland to forest conversion costs may be a major contributor to the lack of reforestation of previously reclaimed mine lands in the Appalachian coal-mining region. Otherwise profitable forestry opportunities may be precluded by these conversion costs, which for many combinations of factors (site class, forest type, timber prices, regeneration intensity, and interest rate) result in negative LEVs. Improved technology and/or knowledge of reforestation practices in these situations may provide opportunities to reduce the costs of converting many of these sites as research continues into these practices.

It also appears that in many cases substantial payments, non-revenue values, or carbon values are required to reach “profitability” under the present circumstances. It is unclear when, or in what form, markets will develop to support any of these add-on values to supplement commercial forestry revenues. However, as these markets do develop, they will only enhance the viability of forestry on reclaimed mined lands, although as we demonstrate in our analysis of carbon payments, the form of the revenue source may itself influence management, potentially mitigating some of the benefits of reforestation.

Introduction

In this Task 4 final report, we (1) examine the financial viability of reforesting reclaimed surface mined lands in the Appalachian coal-mining region, (2) address the economic implications of a shift in reforestation burden from the landowner to the mine operator or government, and (3) explore the opportunities for forest carbon payments on these lands. Our analysis is conducted by comparing land expectation value (LEV) for a range of land use conversion scenarios in which the landowner assumes the costs of site preparation and tree planting, versus scenarios in which the coal company assumes the costs. The comparisons are then used to determine the level of carbon payment required to make forest land use a financially viable option in circumstances where it is not already viable.

Landowner Decision Model

We begin with a model of landowner behavior to describe the reforestation decision, considering a representative landowner who is facing the decision to reforest mined land that was

previously reclaimed into grass cover.¹ Suppose that in the absence of a reforestation option, the landowner possesses an indirect utility function given by,

$$V(R, I, L_0) = V[R(L_0), I] + \phi(L_0) \quad (1)$$

where $V(\cdot)$ is indirect utility which represents maximized landowner welfare when decisions are made optimally, R is the revenue generated from the property (this could be considered land rents and consequently the market value of the land would be included specifically), I is exogenous landowner income, L_0 is the current condition of the property, and Φ measures amenity benefits derived from owning the land. The utility function is assumed to be separable in income-generating activities and amenities, as in nearly all forest economics models (e.g., Hartman 1976 and Faustmann 1849). This assumption is not necessary for our purposes, but it simplifies the discussion below regarding incentives and reforestation of previously mined lands.

The utility function indicates that land condition may be important to landowner welfare in multiple ways, including financial returns (first term on the right hand side (RHS)) and amenities that the landowner derives from the condition of the land (second term on the RHS, which could be thought of as the personal satisfaction or enjoyment from owning and using the land). Moreover, each component of the indirect utility function could, without loss, be expressed as a stream of rents generated from different uses of the land such as grazing, hunting, or production of other revenue-generating activities like carbon storage. That is, we could define the discounted value of annual revenues and amenities occurring over an infinite future from the reclaimed land as:

$$R(L_0) = \int_0^{\infty} a[L_0(z)]e^{-iz} dz; \quad \phi(L_0) = \int_0^{\infty} b[L_0(z)]e^{-iz} dz \quad (2)$$

where i is the interest rate, $a(\cdot)$ are annual revenues generated from the land (perhaps grazing or hunting leases), $b(\cdot)$ are amenity benefits obtained by the landowner each year, z is a variable of integration representing time periods $1, \dots, \infty$, and $L_0(z)$ is land condition at each point in time.

Suppose the landowner represented by (1) is offered an opportunity to reforest previously mined land that was reclaimed into grass cover. Not only will reforestation provide potential harvest revenues when the timber matures, but the stream of amenity benefits may also be different than those identified in (2). The new indirect utility function can be written,

$$V(R, I, L_f) = V[R(L_f), I] + \phi(L_f) \quad (3)$$

where L_f is the condition of the land *after* reforestation (i.e., resulting from the combination of site quality and regeneration intensity), and discounted revenues and amenity benefits are modified as follows:

$$R(L_f) = \frac{pQ_f(t)e^{-it} - c_f e^{-it}}{(1 - e^{-it})} - c_0 + \int_0^{\infty} a[L_f(z)]e^{-iz} dz \quad (4)$$

and

¹ Owners of reclaimed mine lands are quite diverse, including mine operators, land-holding companies, and small private owners. Our representative landowner model is intended to be only a generalized depiction, and all elements of the model may not be applicable to a given owner.

$$\phi(L_f) = \int_0^{\infty} b[L_f(z)]e^{-iz} dz \quad (5)$$

where p is the unit price of timber, $Q_f(t)$ is the volume of timber produced from reforested land at age t (rotation length), c_f is regular reforestation costs after timber harvest, and c_0 is initial reforestation costs incurred when converting grassland into forest.² This formulation is similar to the Hartman (1976) model used commonly in forest landowner analyses (e.g., Amacher *et al.* 2003, Sullivan *et al.* 2005). The first term on the right hand side of (4), including c_0 , is known as land expectation value (LEV). LEV is the present value of benefits from all future forest rotations, beginning with forest establishment. LEV is often called “bare land value.”

Reforestation of reclaimed land improves landowner welfare if:

$$V(R, I, L_f) > V(R, I, L_0) \Leftrightarrow$$

$$V[R(L_f), I] - V[R(L_0), I] + \phi(L_f) - \phi(L_0) > 0 \quad (6)$$

or

$$V \left[\frac{pQ_f(t)e^{-it} - c_f e^{-it}}{(1 - e^{-it})} - c_0 + \int_0^{\infty} a[L_f(z)]e^{-iz} dz, I \right] - V \left[\int_0^{\infty} a[L_0(z)]e^{-iz} dz, I \right]$$

$$+ \int_0^{\infty} b[L_f(z)]e^{-iz} dz - \int_0^{\infty} b[L_0(z)]e^{-iz} dz > 0 \quad (7)$$

Net discounted revenues of timber harvesting in this calculation could be positive or negative, depending on conversion and regular reforestation costs, prices, interest rates and timber volumes produced. Annual revenues and amenity benefits may differ from pre-reforestation values, due to differing land attributes (i.e., L_f versus L_0), though it is difficult to imagine these annual values declining after the land is reforested and the forest starts to grow. It also should be noted that a major component of our calculation below is the conversion cost (c_0) in (4) and (7) above. This undiscounted cost must be paid up front, at the time of reforestation, without any offsetting revenues until the timber reaches maturity in t years. Hence, it is likely to play a major role in determining landowner welfare, the likelihood of reforestation occurring, and the magnitude of additional incentives needed to encourage reforestation on some sites.

In our analysis of the economic aspects of reforesting previously reclaimed mine land that follows, we seek to understand the magnitude of the net discounted revenues under a range of site conditions and reforestation options. The focus of our analysis is the timber revenues and

² This specification of the problem has been simplified for presentation, depicting forest conversion cost as a single, one-time expense. In actuality, there may be multiple costs associated with site conversion (especially the control of established, competing vegetation) that occur over the first few years of initial forest stand establishment. These additional costs are addressed in our empirical analysis.

costs that a landowner faces, due to a paucity of data regarding amenity values depicted in (5). We also evaluate the importance of identifying who pays conversion costs (c_0).

Methods

Growth and Yield

Two forest stand types are considered for reforestation purposes: (1) mixed Appalachian hardwoods on both long and short rotations and (2) white pine. Per-hectare stand volume is estimated using inventory data from reclaimed mined sites and a standard variable-density model of the form:

$$\ln(Q) = b_0 + b_1S + b_2N + b_3 / A + e \quad (8)$$

where volume (Q) is the dependent variable, and site index (S), stand density (N), and age (A^{-1}) are independent variables (Table 4-1). Forest inventory data are taken from a study of reclaimed mined sites (Rodrigue 2001), gathered from 14 planted forest sites on reclaimed land across seven states, in midwestern and eastern U.S. coalfields. The 14 sites, each with an average size of 2.5 ha of uniform and contiguous forest cover, range from 20 to 55 years old, and cover a broad spectrum of mine spoil types.

Table 4-1. Estimated yield coefficients for mixed hardwood and white pine yield (based on data from study by Rodrigue (2001)). Dependent variable is yield (tons).

	Forest Type	
	Mixed Hardwoods	White Pine
Site index (S)		
Coefficient	0.0145**	0.01604**
t-stat	2.90857	3.13600
P-value	0.00529	0.00448
Stand density (N)		
Coefficient	-0.00023	-0.00038
t-stat	-0.86142	-1.50392
P-value	0.39289	0.14565
Age (1/A)		
Coefficient	-11.22905	-34.13108**
t-stat	-1.03875	-3.81126
P-value	0.30364	0.00085
Observations	57	28

* indicates coefficient is significant at $\alpha = 0.10$ level

** indicates coefficient is significant at $\alpha = 0.01$ level

Silvicultural Treatments, Costs, and Site Classes

LEVs are calculated for three levels of reforestation intensity and five different forest site quality classes. It was assumed that silvicultural treatment costs increase with increasing levels of reforestation intensity. The three proposed silvicultural treatments for this study are aimed at sequentially addressing the three major factors limiting reforestation success on reclaimed mined land:

- incompatible ground cover
- soil compaction
- incompatible soil chemical properties and low fertility

Silviculture treatment 1 (low intensity) - This treatment addresses only the existing incompatible ground cover. The following silvicultural activities are assumed: herbicides are used to kill the existing groundcover, trees are planted, and herbicides are used twice in year one and twice in year two to control competing vegetation adjacent to the planted trees.

Silviculture treatment 2 (medium intensity) - This treatment addresses the existing incompatible groundcover and soil compaction problems. The following silvicultural activities are assumed: herbicides are used to kill the existing groundcover, the site is tilled to a depth of 0.7m in one direction to ameliorate soil compaction problems, trees are planted, and herbicides are used twice in year one and twice in year two to control competing vegetation adjacent to the planted trees.

Silviculture treatment 3 (high intensity) – This treatment addresses the existing incompatible groundcover, soil compaction and soil chemical and fertility problems. The following silvicultural activities are assumed: herbicides are used to kill the existing groundcover, the site is tilled to a depth of 0.7m in one direction to ameliorate soil compaction problems, soil chemical and fertility problems are corrected/improved through liming and fertilization, trees are planted, and herbicides are used twice in year one and twice in year two to control competing vegetation adjacent to the planted trees.

The anticipated costs associated with each of the three reforestation activities are estimated from literature (Forest Landowner 2001) and local expert opinion (Table 4-2). Costs of regeneration in future timber rotations are reduced to account for anticipated natural regeneration of future hardwood stands and planting of white pine on uncompacted soils (which do not require additional ripping). Total costs vary by intensity level and species.

Five site classes are considered in our analysis, identified from I to V, with I representing the highest quality site and V representing the lowest (Table 4-3). In addition to influencing timber growth, as indicated in (8), site class is assumed to influence product class (sawtimber v. pulpwood), according to Amacher et al. (1997). On higher quality sites, the proportion of sawtimber tends to be higher, and vice versa on poorer sites (Table 4-4).

Table 4-2. Estimated costs of reforestation activities by year.

	Activity	Cost (\$/ha)	Reforestation Intensity
Year 0	Weed control (herbicide + application)	262.54 ¹	High, medium, low
	Ripping (D8 bulldozer)	296.52 ²	High, medium
	Fertilize (6oz/tree: materials + application)	197.16 ³	High
	Lime (2.47 tons/ha: materials + application)	97.23 ⁴	High
	Hardwood Seedlings (\$0.25/tree)	369.66 ⁵	High, medium, low
	Pine Seedlings (\$0.20/tree)	268.84 ⁵	High, medium, low
	Hand planting	672.11 ⁶	High, medium, low
Year 1	Weed control 1 (herbicide + application)	90.19 ⁷	High, medium, low
	Weed control 2 (herbicide + application)	90.19	High, medium, low
Year 2	Weed control 1 (herbicide + application)	90.19	High, medium, low
	Weed control 2 (herbicide + application)	90.19	High, medium, low
	Fertilize (12 oz/tree)	237.49	High

¹ 1.25 gal/ac @ \$45/gal + application @ \$50/ac (tractor sprayer)

² D8 bulldozer @ \$120/ac

³ \$4/50lbs bag of 10:10:10 + application @ \$7/hr (labor wage)

⁴ \$4.35/ton of lime + application @ \$35/ac (farm tractor)

⁵ 2001 Forest Landowner Nursery Directory (Forest Landowner 2001)

⁶ 1344 trees/ha @ \$0.5/tree

⁷ 0.5gal/ac @ \$45/gal + application @ \$14/ac (hand spray)

Table 4-3. Site index classes for mixed hardwoods and white pine (ft at base age 50).

Site Class	V	IV	III	II	I
mixed hardwoods					
Range	< 51	52-61	62-71	72-80	80 +
Average	46	56	66	75	85
white pine					
Range	< 64	65-77	78-88	89-99	100 +
Average	58	71	83	94	106

Table 4-4. Proportions of sawtimber and pulpwood for various site classes by species (Amacher et al., 1997).

Site Class	Sawtimber	Pulpwood
	----- % -----	
I, II (Hardwoods & Pine)	75	25
III, IV, V (Hardwoods)	66.7	33.4
III, IV, V (Pine)	50	50

Land Expectation Values

Again, LEV is the net present value of forest revenues and costs over an infinite time horizon, argued by economists to be the correct forestry investment criterion because it takes into account both the opportunity cost of money tied up in the investment in forestry over a timber rotation, and the opportunity cost of land (by considering future rotations). We calculate LEV from the perspective of the forest landowner; for which the continuous-time formulation is as follows:³

$$LEV = \frac{pQ_f(t)e^{-it} - c_f e^{-it}}{(1 - e^{-it})} - c_0 \quad (9)$$

To consider the burden of the reforestation costs to the landowner, LEV is calculated with and without the initial conversion cost (c_0).

Both low and high price assumptions are considered in the LEV calculations. Low prices are based on 2003 Timber Mart-South standing timber average prices for mixed hardwoods and pine in the mountains of Virginia and high prices are derived from the 2003 Pennsylvania Woodlands timber market report, with these two locations representing the relative extremes of the timber market in the Appalachian mine region.

In addition, scenarios are developed using 3.5%, 5%, and 7.5% as alternative rates of return (ARR) (variable i in the landowner model). A 5% discount rate is a common baseline for terrestrial carbon studies (e.g., Plantinga et al. 1999; Richards et al. 1993; Stavins 1999) and, as Stavins and Richards (2005, p. 22) note in their recent survey of carbon sequestration work, is a rate that “also represents a social discount rate that has been commonly employed in the analysis of public projects and policies.” The lower and higher rates are included to provide insight into the sensitivity of results to ARR assumptions.

A wide variety of decision parameters must be considered when examining the financial viability of reforesting mined lands, including planting density, regeneration intensity, and rotation age. Unfortunately, our tree growth data do not include detailed product class information, which is essential for meaningful optimization of silvicultural practices, especially in hardwood management. Hence, we do not optimize these management practices for each combination of discount rate, product price level, and site class, but rather examine “typical” practices. Therefore, all LEV calculations are conducted under the assumption of an initial planting density of 1344 stems/ha, representing a standard 10 x 8 ft planting density. Long-rotation hardwood rotation ages were varied from 40 to 80 years, short-rotation hardwood rotation ages were considered between 7 and 20 years, and pine rotation ages were varied between 20 and 40 years.⁴ These rotation age ranges cover the spectrum of typical rotation ages for hardwoods and softwoods, respectively.

³ We present the continuous-time version of LEV here for consistency with our underlying landowner decision model. However, in our analysis we calculate LEV using the discrete-time formulation:

$$LEV = \left[\sum_{t=0}^T R_t / (1+i)^t - \sum_{t=0}^T C_t / (1+i)^t \right] / (1 - (1+i)^{-T}), \text{ where } R_t \text{ and } C_t \text{ are revenue and cost occurring at time } t,$$

respectively, and T is the rotation length.

⁴ Although we consider a range of rotation lengths for both mixed hardwoods and white pine, we only display results for 60-year and 10-year rotations in the long and short rotation hardwoods, respectively, and a 30-year

Results and Discussion

LEVs and the Burden of Reforestation Costs

Of course, the highest LEVs occur on the highest quality sites with the lowest interest rates (Tables 4-5, 4-6, and 4-7). When landowners pay the cost of reforestation, long rotations mixed hardwoods have their highest LEVs at the lowest reforestation intensity level, although those LEVs are only positive when the interest rate is low (3.5%). The relatively long hardwood rotations (compared with white pine and short-rotation hardwoods), combined with substantial up-front reforestation expenses to convert from grasses to a forested condition, make financial viability unlikely. White pine shows its highest LEVs at the medium site intensity level, but these LEVs are only positive on the highest site qualities, and when interest rates are low to moderate. Short-rotation hardwoods LEVs are also only positive at low to moderate interest rates, although positive LEVs can be found even on some of the lower quality sites. The highest LEVs for the short-rotation hardwoods generally occur with the lowest reforestation intensity.

A stark contrast is seen in the calculated LEVs between scenarios where the landowner pays reforestation expenses (c_0) versus a case where the mine operator pays these expenses (Tables 4-5, 4-6, and 4-7).⁵ Although very few combinations of site quality, reforestation intensity, and interest rate led to positive LEVs when landowners paid the reforestation costs, all long-rotation mixed hardwood scenarios (Table 4-5) where the mine operator pays the initial reforestation expenses show positive LEVs, although many are not very large in magnitude, and ultimately may not do much to promote active forest management.

Similar patterns are observed in the white pine and short-rotation hardwood scenarios (Tables 4-6 and 4-7), except that the typically shorter rotation lengths result in more scenarios with positive LEVs when the landowner pays the up-front costs, and yield higher LEVs than long-rotation mixed hardwoods when the mine operator pays the reforestation expenses. The burden of the up-front reforestation expense clearly is a critical consideration for landowners making post-mining land-use decisions.

Not only is the issue of who pays the costs critical in determining profitability, it is also important in determining the appropriate level of reforestation intensity/effort. On both types of hardwood management, the optimal reforestation intensity is the lowest level when the landowner pays those costs, but the optimal intensity is moderate to high when the mine operator pays the upfront costs. Similarly, in the case of white pine, the optimal reforestation intensity is a medium level when the landowner pays the upfront costs, while the optimal intensity is highest when the mine operator pays those costs.

Whether or not it is desirable from a societal standpoint to encourage reforestation of previously reclaimed mined lands depends upon net social values created under reforestation scenarios, with no guarantee that these social values for reforestation are positive. Social benefits, which are not recognized in (3), might include water quality improvement or protection, wildlife habitat improvement, visual quality enhancement, or other public benefits. Social cost

rotation for pine due to a concern for brevity. The other rotations considered show similar patterns to those displayed.

⁵ When considering mine operator expenses for reforestation, we include all of the costs of establishing the stand on the reclaimed mine site in this calculation, including those discounted costs that occur in years subsequent to the initial planting (see Table 2), but do not include costs of establishing future rotations, which would be paid by the landowner.

would be represented by either a government payment made to promote reforestation, or loss in social welfare resulting from a reduction in profits of mine operators who pay for reforestation, depending upon the policy option considered.

It also should be noted that even if net social benefits are positive, we do not consider whether these profit deficits would most appropriately be made by mine operators or the government. Some would interpret the provisions of SMCRA to require mine operators to return to the site and pay the costs of making these lands truly productive, including the establishment of commercial forests where appropriate. However, others would argue that regulatory agencies in the past have not readily accepted or been able to judge the suitability of forest establishment for securing bond release on these reclaimed sites. We leave this question for the policymakers and legal experts who undoubtedly will have to provide the legal interpretation of mine operator responsibilities under SMCRA.

A variety of policy instruments could be designed to remove the burden of forest conversion costs from the landowner, and thereby encourage reforestation. Examples include: lump-sum payments to landowners at the time of conversion, annual payments throughout the rotation, and encouraging/establishing carbon markets in these areas. In the following sections we examine the lump-sum, annual, and carbon payment options further. We consider both payments that allow the landowners to achieve a minimum level of profitability (i.e., $LEV \geq 0$), and those that provide full reimbursement of reforestation expenses.

Table 4-5. Land expectation values of mixed Appalachian hardwood reforestation of reclaimed mine sites when landowner or mine operator pay establishment costs, by site class, reforestation intensity, and discount rate—60-year rotation (\$/ha)

Landowner Pays to Reforest				Mine Operator Pays to Reforest		
5% ARR, low prices						
Site/Intensity	Low	Medium	High	Low	Medium	High
V	-1405.82	-1671.10	-2144.79	200.13	231.36	267.47
IV	-1374.58	-1634.99	-2103.05	231.36	267.47	309.21
III	-1338.47	-1593.25	-2054.79	267.47	309.21	357.47
II	-1275.17	-1520.07	-2001.11	330.77	382.39	411.15
I	-1223.55	-1491.31	-2001.11	382.39	411.15	411.15
5% ARR, high prices						
Site/Intensity	Low	Medium	High	Low	Medium	High
V	-1020.07	-1225.16	-1629.25	585.87	677.30	783.01
IV	-928.64	-1119.45	-1507.05	677.30	783.01	905.21
III	-822.93	-997.25	-1365.77	783.01	905.21	1046.49
II	-625.92	-769.49	-1194.08	980.02	1132.97	1218.18
I	-472.97	-684.28	-1194.08	1132.97	1218.18	1218.18
3.5% ARR, low prices						
Site/Intensity	Low	Medium	High	Low	Medium	High
V	-1098.81	-1315.05	-1738.33	514.39	594.67	687.49
IV	-1018.53	-1222.24	-1631.04	594.67	687.49	794.78
III	-925.72	-1114.95	-1507.00	687.49	794.78	918.82
II	-763.01	-926.84	-1369.01	850.20	982.89	1056.81
I	-630.32	-852.92	-1369.01	982.89	1056.81	1056.81
3.5% ARR, high prices						
Site/Intensity	Low	Medium	High	Low	Medium	High
V	-107.31	-168.81	-413.20	1505.89	1740.92	2012.62
IV	127.71	102.89	-99.09	1740.92	2012.62	2326.73
III	399.41	417.00	264.04	2012.62	2326.73	2689.86
II	905.80	1002.42	705.34	2519.01	2912.15	3131.16
I	1298.94	1221.43	705.34	2912.15	3131.16	3131.16
7.5% ARR, low prices						
Site/Intensity	Low	Medium	High	Low	Medium	High
V	-1547.65	-1836.88	-2328.34	46.77	54.07	62.51
IV	-1540.36	-1828.44	-2318.58	54.07	62.51	72.26
III	-1531.92	-1818.68	-2307.30	62.51	72.26	83.54
II	-1517.12	-1801.58	-2294.76	77.30	89.37	96.09
I	-1505.06	-1794.86	-2294.76	89.37	96.09	96.09
7.5% ARR, high prices						
Site/Intensity	Low	Medium	High	Low	Medium	High
V	-1457.51	-1732.66	-2207.85	136.92	158.29	182.99
IV	-1436.14	-1707.96	-2179.29	158.29	182.99	211.55
III	-1411.44	-1679.40	-2146.28	182.99	211.55	244.56
II	-1365.39	-1626.17	-2106.15	229.03	264.77	284.69
I	-1329.65	-1606.26	-2106.15	264.77	284.69	284.69

Table 4-6. Land expectation values of white pine reforestation of reclaimed mine sites when landowner or mine operator pay establishment costs, by site class, reforestation intensity, and discount rate—30 year rotation (\$/ha).

Landowner Pays to Reforest				Operator Pays to Reforest		
				5% ARR, low prices		
Site/Intensity	Low	Medium	High	Low	Medium	High
V	-1090.77	-1193.86	-1593.36	447.97	641.40	751.70
IV	-879.47	-937.70	-1282.82	659.27	897.56	1062.24
III	-641.18	-648.83	-932.62	897.56	1186.43	1412.44
II	71.39	215.02	114.61	1610.13	2050.28	2459.67
I	511.54	748.61	422.49	2050.28	2583.87	2767.55
				5% ARR, high prices		
Site/Intensity	Low	Medium	High	Low	Medium	High
V	-891.79	-952.63	-1300.92	646.95	882.63	1044.14
IV	-634.35	-640.54	-922.57	904.39	1194.72	1422.49
III	-344.02	-288.58	-495.89	1194.72	1546.68	1849.17
II	459.49	685.51	684.98	1998.23	2520.77	3030.04
I	982.03	1318.98	1050.49	2520.77	3154.24	3395.55
				3.5% ARR, low prices		
Site/Intensity	Low	Medium	High	Low	Medium	High
V	-726.40	-667.28	-984.05	819.61	1175.25	1374.57
IV	-337.91	-196.32	-413.10	1208.09	1646.21	1945.51
III	100.20	334.81	230.77	1646.21	2177.34	2589.39
II	1410.33	1923.06	2156.20	2956.34	3765.59	4514.81
I	2219.58	2904.11	2722.26	3765.59	4746.64	5080.88
				3.5% ARR, high prices		
Site/Intensity	Low	Medium	High	Low	Medium	High
V	-360.55	-223.76	-446.38	1185.45	1618.76	1912.24
IV	112.77	350.05	249.25	1658.78	2192.58	2607.86
III	646.57	997.16	1033.74	2192.58	2839.69	3392.35
II	2123.88	2788.10	3204.87	3669.89	4630.62	5563.49
I	3084.62	3952.79	3876.89	4630.62	5795.31	6235.51
				7.5% ARR, low prices		
Site/Intensity	Low	Medium	High	Low	Medium	High
V	-1333.85	-1547.51	-1998.88	193.38	276.24	324.77
IV	-1243.33	-1437.78	-1865.86	283.89	385.97	457.79
III	-1141.26	-1314.03	-1715.84	385.97	509.71	607.80
II	-836.02	-943.99	-1267.24	691.21	879.75	1056.40
I	-647.47	-715.42	-1135.36	879.75	1108.32	1188.28
				7.5% ARR, high prices		
Site/Intensity	Low	Medium	High	Low	Medium	High
V	-1248.61	-1444.17	-1873.61	278.62	379.57	450.03
IV	-1138.33	-1310.48	-1711.54	388.89	513.26	612.10
III	-1013.96	-1159.72	-1528.76	513.26	664.03	794.88
II	-669.77	-742.45	-1022.92	857.45	1081.29	1300.72
I	-445.93	-471.10	-866.35	1081.29	1352.65	1457.29

Table 4-7. Land expectation values of short-rotation mixed Appalachian hardwood reforestation of reclaimed mine sites when landowner or mine operator pay establishment costs, by site class, reforestation intensity, and discount rate—10 year rotation (\$/ha).

Landowner Pays to Reforest				Mine Operator Pays to Reforest		
5% ARR, low prices						
Site/Intensity	Low	Medium	High	Low	Medium	High
V	-951.19	-1145.52	-1537.19	654.75	756.94	875.07
IV	-849.00	-1027.39	-1400.62	756.94	875.07	1011.64
III	-730.87	-890.82	-1242.73	875.07	1011.64	1169.53
II	-608.86	-749.77	-1172.88	997.08	1152.69	1239.38
I	-453.25	-663.08	-1172.88	1152.69	1239.38	1239.38
5% ARR, high prices						
Site/Intensity	Low	Medium	High	Low	Medium	High
V	-331.64	-429.28	-709.17	1274.30	1473.18	1703.09
IV	-132.76	-199.37	-443.37	1473.18	1703.09	1968.89
III	97.15	66.43	-136.08	1703.09	1968.89	2276.18
II	334.61	340.94	-0.14	1940.55	2243.40	2412.12
I	637.46	509.66	-0.14	2243.40	2412.12	2412.12
3.5% ARR, low prices						
Site/Intensity	Low	Medium	High	Low	Medium	High
V	-610.36	-750.36	-1085.51	1002.85	1159.36	1340.30
IV	-453.84	-569.42	-876.33	1159.36	1340.30	1549.48
III	-272.90	-360.24	-634.51	1340.30	1549.48	1791.31
II	-86.03	-144.21	-527.52	1527.17	1765.52	1898.30
I	152.31	-11.43	-527.52	1765.52	1898.30	1898.30
3.5% ARR, high prices						
Site/Intensity	Low	Medium	High	Low	Medium	High
V	338.58	346.67	182.73	1951.78	2256.40	2608.55
IV	643.19	698.82	589.84	2256.40	2608.55	3015.66
III	995.34	1105.93	1060.50	2608.55	3015.66	3486.31
II	1359.04	1526.39	1268.72	2972.24	3436.12	3694.53
I	1822.91	1784.81	1268.72	3436.12	3694.53	3694.53
7.5% ARR, low prices						
Site/Intensity	Low	Medium	High	Low	Medium	High
V	-1206.34	-1442.29	-1872.17	388.08	448.65	518.67
IV	-1145.77	-1372.27	-1791.22	448.65	518.67	599.62
III	-1075.75	-1291.32	-1697.64	518.67	599.62	693.20
II	-1003.44	-1207.72	-1656.24	590.99	683.22	734.60
I	-911.20	-1156.34	-1656.24	683.22	734.60	734.60
7.5% ARR, high prices						
Site/Intensity	Low	Medium	High	Low	Medium	High
V	-839.12	-1017.76	-1381.38	755.30	873.18	1009.46
IV	-721.24	-881.49	-1223.84	873.18	1009.46	1167.00
III	-584.97	-723.94	-1041.71	1009.46	1167.00	1349.14
II	-444.22	-561.23	-961.13	1150.20	1329.71	1429.71
I	-264.71	-461.23	-961.13	1329.71	1429.71	1429.71

Lump-Sum and Annual Payments to Landowners

One potential policy option for encouraging reforestation of previously reclaimed surface mine sites is to provide a lump-sum payment to the landowner at the time of conversion. A payment could be established that provides the minimum level of profitability, in which case the payment would be set at the absolute value of those LEVs that are less than zero (Table 4-8). This payment would vary by site quality and regeneration intensity, and would be zero for cases where LEV is already greater than or equal to zero.⁶ Alternatively, a payment level could also be established to provide full reimbursement of reforestation expenses, in which case the payment would be set at the present value of the initial reforestation costs (Table 4-9), which varies by regeneration intensity, but not by site class:⁷

Calculated minimum profitability lump-sum payments (Table 4-8) vary between zero, for those scenarios with estimated LEVs already greater than zero in which interest rates are low and site productivity is high, and over \$2300 per hectare for long-rotation mixed hardwood scenarios with high interest rates and low site productivity. It is interesting that although sort-rotation hardwood payments less than the corresponding long-rotation payments, they alternate with the required white pine payments, with white pine requiring a smaller payment for profitability on better sites, and short-rotation hardwoods requiring a lower payment on poor sites.

By adjusting regeneration intensity as appropriate for site productivity and market situation, the payment required for minimum profitability can be reduced somewhat, as compared with some of the highest payment scenarios. Overall, our results illustrate that smaller investments typically should be made in sites with poor productivity where improved growth does not offset the additional cost, or in situations where high alternative rates of return result in greater discounting of the future benefits of those investments.

Lump-sum payments required for full reimbursement of forest conversion expenses range from \$1527 per hectare for white pine under a high ARR and low reforestation intensity, up to nearly \$2426 per hectare for mixed hardwoods under a low ARR and high intensity reforestation (Table 4-9). The fact that reforestation cost is higher for hardwoods than for pines, and that it increases with increasing intensity, is evident in the uniformly higher payment required for hardwood conversion compared with pine conversion, and in the higher payment required for reimbursement of higher intensity effort. With regard to the lower payments occurring under a higher ARR, these results seem counterintuitive. However, the costs occurring in the later years of reforestation (i.e., second and third years) are discounted to a greater degree at higher interest rates, therefore requiring a lower payment in the first year than in situations with a lower ARR. Recall that under this scheme, productivity and growth do not matter, because the payment is based only on recovery of the cost of reforesting.

⁶ Minimum profitability payment = $\left| LEV_{jk} \right| \forall LEV_{jk} \leq 0$, and 0 otherwise, where LEV_{jk} is land expectation value for site class j and regeneration intensity k prior to any payments made to encourage reforestation.

⁷ Full reimbursement payment = $\sum_{t=0}^R c_{0kt} / (1+i)^t$ where c_{0kt} is reforestation cost for regeneration intensity k

occurring at time t , and R is the time at which reforestation is fully established and bond release is achieved. Calculated full reimbursement payments do not vary in our study by site class, because costs do not vary across site classes for a given forest type and regeneration intensity level. Higher intensity regeneration simply costs more, and improved growth is not a factor in this payment option.

Table 4-8. Lump-sum payment required at reforestation inception to provide minimal profitability (LEV $\geq$ 0) for landowner of reclaimed mine land, by site class, reforestation intensity, and discount rate (\$/ha).

Mixed Hardwoods (60-year rotation)				White Pine (30-year rotation)			Short-Rotation Hardwoods (10-year rotation)		
5% ARR, low prices									
Site/Intensity	Low	Medium	High	Low	Medium	High	Low	Medium	High
V	1405.82	1671.10	2144.79	1090.77	1193.86	1593.36	951.19	1145.52	1537.19
IV	1374.58	1634.99	2103.05	879.47	937.70	1282.82	849.00	1027.39	1400.62
III	1338.47	1593.25	2054.79	641.18	648.83	932.62	730.87	890.82	1242.73
II	1275.17	1520.07	2001.11	*	*	*	608.86	749.77	1172.88
I	1223.55	1491.31	2001.11	*	*	*	453.25	663.08	1172.88
5% ARR, high prices									
Site/Intensity	Low	Medium	High	Low	Medium	High	Low	Medium	High
V	1020.07	1225.16	1629.25	891.79	952.63	1300.92	331.64	429.28	709.17
IV	928.64	1119.45	1507.05	634.35	640.54	922.57	132.76	199.37	443.37
III	822.93	997.25	1365.77	344.02	288.58	495.89	*	*	136.08
II	625.92	769.49	1194.08	*	*	*	*	*	0.14
I	472.97	684.28	1194.08	*	*	*	*	*	0.14
3.5% ARR, low prices									
Site/Intensity	Low	Medium	High	Low	Medium	High	Low	Medium	High
V	1098.81	1315.05	1738.33	726.40	667.28	984.05	610.36	750.36	1085.51
IV	1018.53	1222.24	1631.04	337.91	196.32	413.10	453.84	569.42	876.33
III	925.72	1114.95	1507.00	*	*	*	272.90	360.24	634.51
II	763.01	926.84	1369.01	*	*	*	86.03	144.21	527.52
I	630.32	852.92	1369.01	*	*	*	*	11.43	527.52
3.5% ARR, high prices									
Site/Intensity	Low	Medium	High	Low	Medium	High	Low	Medium	High
V	107.31	168.81	413.20	360.55	223.76	446.38	*	*	*
IV	*	*	99.09	*	*	*	*	*	*
III	*	*	*	*	*	*	*	*	*
II	*	*	*	*	*	*	*	*	*
I	*	*	*	*	*	*	*	*	*
7% ARR, low prices									
Site/Intensity	Low	Medium	High	Low	Medium	High	Low	Medium	High
V	1547.65	1836.88	2328.34	1333.85	1547.51	1998.88	1206.34	1442.29	1872.17
IV	1540.36	1828.44	2318.58	1243.33	1437.78	1865.86	1145.77	1372.27	1791.22
III	1531.92	1818.68	2307.30	1141.26	1314.03	1715.84	1075.75	1291.32	1697.64
II	1517.12	1801.58	2294.76	836.02	943.99	1267.24	1003.44	1207.72	1656.24
I	1505.06	1794.86	2294.76	647.47	715.42	1135.36	911.20	1156.34	1656.24
7% ARR, high prices									
Site/Intensity	Low	Medium	High	Low	Medium	High	Low	Medium	High
V	1457.51	1732.66	2207.85	1248.61	1444.17	1873.61	839.12	1017.76	1381.38
IV	1436.14	1707.96	2179.29	1138.33	1310.48	1711.54	721.24	881.49	1223.84
III	1411.44	1679.40	2146.28	1013.96	1159.72	1528.76	584.97	723.94	1041.71
II	1365.39	1626.17	2106.15	669.77	742.45	1022.92	444.22	561.23	961.13
I	1329.65	1606.26	2106.15	445.93	471.10	866.35	264.71	461.23	961.13

* indicates LEV is positive without additional payments

Table 4-9. Lump-sum payment required at reforestation inception to provide full reimbursement of forest conversion costs to landowner of reclaimed mine land, reforestation intensity, and discount rate (\$/ha).

Forest Type/Intensity	Low	Medium	High
5% ARR			
Mixed hardwoods (10- and 60-year rotations)	1605.94	1902.46	2412.26
White pine (30-year rotation)	1538.74	1835.26	2345.06
3.5% ARR			
Mixed hardwoods	1613.21	1909.73	2425.82
White pine	1546.01	1842.53	2358.62
7.5% ARR			
Mixed hardwoods	1594.42	1890.94	2390.84
White pine	1527.22	1823.74	2323.64

The one-time, lump-sum payments discussed above are also converted to annual payments made over the first timber rotation,⁸ for both the minimum profitability level (Table 4-10) and the full cost reimbursement level (Table 4-11). In the case of annualized, minimum profitability payments (Table 4-10), the required payments range from zero to over \$266 per hectare per year, but now some long-rotation hardwood payments are close to, or even lower than, annual payments required to make white pine conversion profitable. This finding is in contrast to many of the lump-sum payments required to make reforestation of these previously reclaimed mine lands minimally profitable, a result of the longer rotation lengths over which the annual hardwood payments are amortized. The required short-rotation hardwood payments often are highest due to only having a 10-year rotation over which to amortize the minimum profitability payments. In the case of annualized, full cost reimbursement payments (Table 4-11), payments range from just under \$65 per hectare per year to nearly \$197 per hectare per year, and in every scenario the required mixed hardwood payment is lower than its white pine counterpart—again due to the longer rotation length for hardwoods.

⁸ The discrete annual payment was calculated as: $a = \frac{P \cdot i \cdot (1+i)^t}{(1+i)^t - 1}$, where P is the lump-sum payment. Our

analysis is based on the assumption that this payment would begin at the time of reforestation and conclude at the time of harvest at the end of the first rotation of timber.

Table 4-10. Annual payment required over one rotation to allow minimal profitability (LEV ≥ 0) for landowner of reclaimed mine land, by site class, reforestation intensity, and discount rate (\$/ha/yr).

Mixed Hardwoods (60-year rotation)				White Pine (30-year rotation)			Short-Rotation Hardwoods (10-year rotation)		
5% ARR, low prices									
Site/Intensity	Low	Medium	High	Low	Medium	High	Low	Medium	High
V	74.27	88.28	113.31	70.96	77.66	103.65	123.18	148.35	199.07
IV	72.62	86.37	111.10	57.21	61.00	83.45	109.95	133.05	181.39
III	70.71	84.17	108.55	41.71	42.21	60.67	94.65	115.37	160.94
II	67.36	80.30	105.72	*	*	*	78.85	97.10	151.89
I	64.64	78.78	105.72	*	*	*	58.70	85.87	151.89
5% ARR, high prices									
Site/Intensity	Low	Medium	High	Low	Medium	High	Low	Medium	High
V	53.89	64.72	86.07	58.01	61.97	84.63	42.95	55.59	91.84
IV	49.06	59.14	79.61	41.27	41.67	60.01	17.19	25.82	57.42
III	43.47	52.68	72.15	22.38	18.77	32.26	*	*	17.62
II	33.07	40.65	63.08	*	*	*	*	*	0.02
I	24.99	36.15	63.08	*	*	*	*	*	0.02
3.5% ARR, low prices									
Site/Intensity	Low	Medium	High	Low	Medium	High	Low	Medium	High
V	44.05	52.72	69.69	39.50	36.28	53.50	73.39	90.22	130.52
IV	40.83	49.00	65.39	18.37	10.67	22.46	54.57	68.47	105.37
III	37.11	44.70	60.41	*	*	*	32.81	43.32	76.29
II	30.59	37.16	54.88	*	*	*	10.34	17.34	63.43
I	25.27	34.19	54.88	*	*	*	*	1.37	63.43
3.5% ARR, high prices									
Site/Intensity	Low	Medium	High	Low	Medium	High	Low	Medium	High
V	4.30	6.77	16.56	19.60	12.17	24.27	*	*	*
IV	*	*	3.97	*	*	*	*	*	*
III	*	*	*	*	*	*	*	*	*
II	*	*	*	*	*	*	*	*	*
I	*	*	*	*	*	*	*	*	*
7% ARR, low prices									
Site/Intensity	Low	Medium	High	Low	Medium	High	Low	Medium	High
V	110.24	130.84	165.85	107.49	124.71	161.08	171.76	205.35	266.55
IV	109.72	130.24	165.15	100.20	115.87	150.36	163.13	195.38	255.03
III	109.12	129.54	164.35	91.97	105.89	138.27	153.16	183.85	241.71
II	108.06	128.33	163.45	67.37	76.07	102.12	142.87	171.95	235.81
I	107.20	127.85	163.45	52.18	57.65	91.49	129.73	164.64	235.81
7% ARR, high prices									
Site/Intensity	Low	Medium	High	Low	Medium	High	Low	Medium	High
V	103.82	123.42	157.26	100.62	116.38	150.99	119.47	144.91	196.68
IV	102.30	121.66	155.23	91.73	105.61	137.93	102.69	125.50	174.25
III	100.54	119.62	152.88	81.71	93.46	123.20	83.29	103.07	148.32
II	97.26	115.83	150.02	53.97	59.83	82.43	63.25	79.91	136.84
I	94.71	114.41	150.02	35.94	37.96	69.82	37.69	65.67	136.84

* indicates LEV is positive without additional payments

Table 4-11. Annual payment required over one rotation to provide full reimbursement of forest conversion costs to landowner of reclaimed mine land, reforestation intensity, and discount rate (\$/ha/yr).

Forest Type/Intensity	Low	Medium	High
5% ARR			
Mixed hardwoods (10- and 60-year rotations)	84.84	100.50	127.44
White pine (30-year rotation)	100.10	119.39	152.55
3.5% ARR			
mixed hardwoods	64.67	76.56	97.25
white pine	84.06	100.18	128.24
7.5% ARR			
mixed hardwoods	121.16	143.70	181.68
white pine	129.31	154.42	196.75

Even though these payments could generate positive LEVs or even provide full reimbursement of grassland to forest conversion costs, there is no guarantee that the payments will actually cause landowners to reforest in practice. Equation (7) indicates that it is landowner *utility* associated with forestland profitability that will be the determining factor in actual conversion—utility that likely would include cash flow timing, amenities, and even the credit position of the landowner. Our calculated values could be viewed as the minimum levels of payments required to cause grassland to forest conversion on reclaimed mine lands in the Appalachian mine region.

Further, it should be noted that we do not determine whether any of these types of reforestation payments should come from the mine operators or the government. Some would interpret the provisions of SMCRA to require mine operators to go back and pay the costs of making these lands truly productive, including the establishment of commercial forests where appropriate. However, others would argue that regulatory agencies in the past have not readily accepted or been able to judge the suitability of forest establishment for securing bond release on these reclaimed sites. We leave this question for the policymakers and legal experts who undoubtedly will have to provide the legal interpretation of mine operator responsibilities under SMCRA.

Carbon Payments

In this section we examine the magnitude of carbon payments necessary for financial viability. Although a carbon payment is not policy instrument, in and of itself, promoting the establishment of carbon markets could provide a valuable revenue source for forest owners, including those who own reforested mine lands. The body of research directed toward the question of the cost of sequestering carbon through forest establishment in a variety of land-use situations is extensive, though much of it is perhaps not directly relevant to previously reclaimed mine lands (examples include Moulton and Richards 1990, Adams et al. 1993, Richards et al. 1993, Hoen and Solberg 1994, Parks and Hardie 1995, Sedjo et al. 1995, Plantinga et al. 1999, Stavins 1999, van Kooten et al. 2000, and van 't Veld and Plantinga 2005). In the context of our study, rather than asking how much it would cost to sequester carbon on reclaimed mined lands,

we ask how much carbon would have to be worth to make forest conversion viable on reclaimed mine sites. In answering this question, we consider two forms of a carbon payment: (1) an annual payment based upon the accumulated carbon on the site (up to the year of the payment), and (2) an annual payment based upon the incremental carbon added to the site each year. The former approach would reward a forest landowner who retains carbon on-site, perhaps reflecting a concern that carbon removed from a site could return quickly to the atmosphere. The latter approach would reward a landowner for keeping the carbon accumulation rate high, and might be appropriate where carbon removed from a site would be expected to be bound up largely in durable products, such as wooden structures.

Considering first a carbon payment based upon accumulated site carbon, we determine the level of payment required for a minimal level of profitability from forestry by setting the LEV of forestry plus carbon production equal to zero, and solving for s_{ca} , the annual payment per ton of accumulated carbon found on site each year, in (10),

$$R(L_f) = \frac{pQ_f(t)e^{-it} - c_f e^{-it}}{(1 - e^{-it})} - c_0 + s_{ca} \int_0^{\infty} x(z)e^{-iz} dz = 0 \quad (10)$$

where $x(z)$ is the quantity of carbon accumulated on the forested site at time z , which could include both above-ground carbon sequestered in the timber at a given time and carbon sequestered in the soil. This formulation is based on an assumption that timber harvesting will occur periodically as a part of optimal management of the land, rather than some other models used to consider forest carbon which assume that land is converted to permanent, unharvested forest (e.g., van 't Veld and Plantinga 2005).

Conversion from timber volume to total carbon content ($x(z)$) in equation (10) is based on conversion factors used by van Kooten et al. (2000). Their conversion involves estimating total above-ground and root biomass using a set of hardwood and softwood-specific biomass factors, and then converting total biomass to carbon content. Given the difficulty of assessing soil carbon sequestration rates over the life of a stand, and across rotations, we focus on the carbon held in standing timber and the roots.

Calculated payments required for achieving a minimal level of profitability under the accumulated carbon payment approach (Table 4-12) range from zero dollars per ton per year in white pine plantations with low to moderate interest rates and moderate to high productivity levels (where LEVs already are greater than zero), to \$11.60 per ton per year for short-rotation hardwoods in situations with a high interest rate and low productivity. Interestingly, one forest type does not show uniform dominance over the other in terms of requiring the lowest (or highest) carbon payment. On the best quality sites under low to moderate interest rates, required carbon payments are lowest for white pine, whereas long-rotation mixed hardwoods require the lowest payments on the poorest sites. These findings are a consequence of differing relative growth rates between pines and hardwoods (which are dependent upon site quality), and rotation length (where long-rotation hardwoods carry higher volumes of timber and carbon longer before starting the next rotation).

Table 4-12. Annual payment required per ton of cumulative carbon stored on-site to allow minimal profitability (LEV ≥ 0) for landowner of reclaimed mine land, by site class, reforestation intensity, and discount rate (\$/ton/yr)

	Mixed Hardwoods (60-year rotation)			White Pine (30-year rotation)			Short-Rotation Hardwoods (10-year rotation)		
5% ARR, low prices									
Site/Intensity	Low	Medium	High	Low	Medium	High	Low	Medium	High
V	2.25	2.34	2.61	4.00	3.61	3.97	4.84	5.10	5.99
IV	1.92	1.99	2.23	2.62	2.30	2.60	3.78	4.00	4.77
III	1.63	1.69	1.90	1.57	1.31	1.56	2.85	3.04	3.70
II	1.37	1.43	1.75	*	*	*	2.10	2.26	3.31
I	1.15	1.31	1.75	*	*	*	1.37	1.87	3.31
3.5% ARR, low prices									
Site/Intensity	Low	Medium	High	Low	Medium	High	Low	Medium	High
V	1.10	1.14	1.32	1.64	1.24	1.51	2.10	2.26	2.86
IV	0.89	0.93	1.08	0.62	0.30	0.51	1.37	1.50	2.02
III	0.70	0.74	0.87	*	*	*	0.72	0.83	1.28
II	0.51	0.54	0.75	*	*	*	0.20	0.29	1.01
I	0.37	0.46	0.75	*	*	*	*	0.02	1.01
7.5% ARR, low prices									
Site/Intensity	Low	Medium	High	Low	Medium	High	Low	Medium	High
V	4.52	4.68	5.17	9.21	8.81	9.39	9.76	10.22	11.60
IV	3.92	4.06	4.49	6.97	6.65	7.11	8.12	8.50	9.71
III	3.40	3.53	3.90	5.27	5.01	5.40	6.67	7.00	8.04
II	2.98	3.09	3.67	3.24	3.02	3.34	5.51	5.80	7.43
I	2.58	2.87	3.67	2.07	1.89	2.72	4.38	5.19	7.43

* indicates LEV is positive without additional payments.

A carbon payment based upon incremental carbon volume, rather than accumulated volume, is calculated similarly by solving for s_{cg} in:

$$R(L_f) = \frac{pQ_f(t)e^{-it} - c_f e^{-it}}{(1 - e^{-it})} - c_0 + s_{cg} \int_0^{\infty} x(z)e^{-iz} dz = 0 \quad (11)$$

Note that $x(z)$, the cumulative on-site production of carbon, in (10) is replaced by $x(z)$, the annual carbon increment in (11), or $x(z) = dx/dt$. That is, the annual payment s_{cg} in (11) will be based only on the carbon grown on the site that year.⁹

Payments required to make reforestation viable when incremental carbon is considered (Table 4-13) are substantially larger than those found for accumulated carbon, reaching over \$83/ton for white pine grown on the poorest sites with the highest interest rate and site preparation intensity. The larger magnitude of this form of payment may be somewhat

⁹ This formulation is similar to the model that Englin and Callaway (1993) use to examine the effects of climate change on optimal forest management under a pre-defined set of possible carbon payments, except that we do not attempt to value harvested carbon, which would require additional assumptions about end use and carbon release rate.

deceptive, however, as this payment occurs only once at the time that the carbon is sequestered, whereas under the accumulated carbon scheme, payment occurs annually for a given ton, up until the time of harvest. The pattern of incremental carbon payments (Table 4-13) shows that white pine payments are lower for higher quality sites, while short-rotation hardwood payments are lowest on poor quality sites. Incremental carbon payments required to make long-rotation hardwoods profitable are not lowest under any circumstances. Again, these findings reflect a combination of factors that include relative growth rates of hardwoods versus pines (and consequently, the rate of carbon accumulation), the differing rotation lengths, and the time value of money.

Table 4-13. Payment required per ton of carbon added to the site that allows minimal profitability ($LEV \geq 0$) for landowner of reclaimed mine land, by site class, reforestation intensity, and discount rate (\$/ton/yr).

Mixed Hardwoods (60-year rotation)				White Pine (30-year rotation)			Short-Rotation Hardwoods (10-year rotation)		
5% ARR, low prices									
Site/Intensity	Low	Medium	High	Low	Medium	High	Low	Medium	High
V	42.53	44.11	49.36	39.15	35.34	38.91	20.48	21.56	25.27
IV	36.28	37.63	42.19	25.62	22.53	25.43	15.98	16.89	20.11
III	30.81	31.97	35.93	15.41	12.86	15.25	12.02	12.79	15.57
II	25.94	26.95	33.11	*	*	*	8.86	9.53	13.92
I	21.69	24.67	33.11	*	*	*	5.76	7.87	13.92
3.5% ARR, low prices									
Site/Intensity	Low	Medium	High	Low	Medium	High	Low	Medium	High
V	25.80	26.92	31.03	16.99	12.87	15.66	9.05	9.72	12.29
IV	20.85	21.82	25.37	6.42	3.07	5.34	5.88	6.44	8.66
III	16.52	17.34	20.42	*	*	*	3.09	3.56	5.47
II	12.03	12.74	17.56	*	*	*	0.86	1.26	4.31
I	8.66	10.94	17.56	*	*	*	*	0.09	4.31
7.5% ARR, low prices									
Site/Intensity	Low	Medium	High	Low	Medium	High	Low	Medium	High
V	62.76	65.01	71.88	81.68	78.17	83.29	40.09	41.90	47.52
IV	54.51	56.45	62.42	61.81	58.96	63.11	33.29	34.83	39.71
III	47.30	48.97	54.15	46.80	44.45	47.87	27.31	28.63	32.86
II	41.41	42.87	50.98	28.74	26.76	29.64	22.55	23.69	30.36
I	35.81	39.87	50.98	18.36	16.73	24.12	17.88	21.19	30.36

* indicates LEV is positive without additional payments

Even though payment levels calculated here could generate positive LEVs, there is no guarantee that the availability of carbon payments at these levels will actually cause landowners to reforest in practice. Equation (7) indicates that it is landowner *utility* associated with forestland profitability that will be the determining factor in actual conversion—utility that likely would include cash flow timing, amenities, and even the wealth and credit position of the landowner. Our calculated values could be viewed as the minimum levels of payments required to cause grassland to forest conversion on reclaimed mine lands in the Appalachian mine region.

Greater understanding of the role of issues such as cash flows and landowner credit are needed, however, for a full picture of the potential success of any program to encourage reforestation.

It is also difficult to determine at this point whether our calculated carbon payments are reasonable. Sufficient experience with carbon trading markets for timber has not been gained to conclude with any certainty that these values could be achieved under the circumstances found on reclaimed mine sites. However, our calculated carbon payments do provide a basis for understanding the role that carbon trading could play in enhancing the financial viability of forestry on reclaimed mine sites as these markets develop more fully.

At this point it is unclear how carbon markets will work for forest-based carbon on a large scale, and in particular the specifics of how payments might be structured are unknown. Certainly, payments for general land use changes that do not recognize actual measures of carbon storage will not adequately reward management contributions, and thus may have little effect on terrestrial carbon sequestration efforts. In our analysis, we have considered two payment schemes that are based on carbon grown or accumulated on a forested site. However, these schemes may provide substantially different incentives. It might be anticipated that an annual payment based upon accumulated carbon would encourage a longer rotation length that would keep the carbon on-site for a longer period of time, while the payment based upon annual growth of carbon could generate an incentive to shorten the rotation length in order to keep the timber and carbon stock growing closer to the highest rate. We examine these contentions in the following section using comparative statics analyses of forest carbon payments on reclaimed sites.

Comparative Statics of Forest Carbon Payments: Accumulated v. Incremental Carbon Payments

To determine the effect of a carbon payment on stand management, we consider the revenue component of our landowner decision model, which includes a carbon payment, and assume that a landowner seeks to maximize this function:

$$R(L_f) = \frac{pQ_f(t)e^{-it} - c_f e^{-it} - c_0(1 - e^{-it}) + s_{ca} \int_0^t x(z)e^{-iz} dz}{(1 - e^{-it})} \quad (12)$$

This form of the decision model incorporates an annual carbon payment that is based upon the total accumulated carbon stored on the forested site. Results for a payment based upon annual increments of carbon growth are derived in a parallel fashion by substituting $x(t)$ for $x(z)$ in (12). Further, for tractability we have modified the revenue portion of the decision model from (7) by assuming that carbon volume is a direct function of stand age, and thus carbon accumulates in an identical fashion through each of an infinite series of timber rotations.

The first order necessary condition for finding a maximum of (12) is determined to be as follows:

$$\frac{\partial R}{\partial t} = pQ'_f(t) + s_{ca}x(t) - ipQ_f(t) + ic_f - ic_0 - iR = 0 \quad (13)$$

The second order sufficiency condition is then:

$$\frac{\partial^2 R}{\partial t^2} = pQ_f''(t) + s_{ca}x'(t) - ipQ_f'(t) < 0 \quad (14)$$

Taking the total derivative of (13) yields:

$$\begin{aligned} & \left[Q_f'(t) - iQ_f(t) - i \frac{Q_f(t)e^{-it}}{(1-e^{-it})} \right] dp + \left[i + \frac{e^{-it}}{(1-e^{-it})} \right] dc_f + \left[-pQ(t) + c_f - c_0 - i \frac{dR}{di} - R \right] di \\ & + [pQ_f''(t) + s_{ca}x'(t) - ipQ_f'(t)]dt + \left[x(t) - i \frac{\int_0^t x(z)e^{-iz} dz}{(1-e^{-it})} \right] ds_{ca} = 0 \end{aligned}$$

Finally, using the implicit function theorem, we can identify the relationship between the optimized rotation length and carbon payment:

$$\frac{dt}{ds_{ca}} = - \frac{\left[x(t) - i \frac{\int_0^t x(z)e^{-iz} dz}{(1-e^{-it})} \right]}{[pQ_f''(t) + s_{ca}x'(t) - ipQ_f'(t)]} \geq 0 \quad (15)$$

The denominator of (15) is negative according to the second order sufficiency condition for a maximum. As long as carbon volume is an increasing function of stand age, the numerator is necessarily positive, as the terminal cumulative carbon quantity $x(t)$ would be greater than what amounts to a weighted average quantity across time (time weighted by the discount rate). Hence, the overall sign of (15) is positive, implying that optimal rotation length will increase with an increasing carbon payment of this form.

Similarly, a condition can be derived to explain the direction of change in optimal rotation length for a change in a carbon payment that is based upon annual increments of carbon:

$$\frac{dt}{ds_{cg}} = - \frac{\left[\mathfrak{x}(t) - i \frac{\int_0^t \mathfrak{x}(z)e^{-iz} dz}{(1-e^{-it})} \right]}{[pQ_f''(t) + s_{cg}\mathfrak{x}'(t) - ipQ_f'(t)]} \begin{matrix} \leq \\ \geq \end{matrix} 0 \quad (16)$$

In this case, the condition cannot be signed definitively. While the terminal rate of carbon growth at harvest $\mathfrak{x}(t)$ is greater than what amounts to the average rate of growth, the numerator will be positive, making (16) positive, and an increase in the carbon payment will extend the rotation. However, at optimized rotation lengths in which the terminal growth rate of carbon has

fallen below the average, (16) will be negative, and an increase in the incremental carbon payment would shorten the rotation length.

Empirical computation of the switching point in (16) is possible, and based on our carbon calculations, this point occurs at age 12 for hardwoods on all site class, interest rate, and regeneration intensity combinations. Hence, for any optimized hardwood rotation length beyond 12 years, we would anticipate that an increase in the incremental carbon payment per ton would shorten the rotation length. The switch between a positive and negative value for (16) occurred for white pine between the ages of 40 and 44, depending on site quality, interest rate, and regeneration combination. For most white pine rotations, therefore, we would expect the optimal rotation length to be extended with an increase in the incremental carbon payment.

In summary, we find: (1) that under an annual payment based upon accumulated carbon, the rotation length of the stand increases unambiguously as payment increases, and 2) that increases in an annual payment based upon carbon increment may lead to a shortening of rotation length under some circumstances. A shorter rotation length would mean that carbon is removed from the site sooner, which could be undesirable if the carbon were expected to be released quickly back into the atmosphere (i.e., sequestered into products with a short carbon half-life), although it could also be desirable if the wood is transformed into durable products with a long carbon half-life. These findings suggest that carbon payment structure should be considered carefully if incentives for potentially adverse management responses are to be avoided.

Conclusions

Based on our results, it seems clear that grassland-to-forest conversion costs may be a major contributor to the lack of reforestation of previously reclaimed mine lands in the Appalachian coal-mining region. Otherwise profitable forestry opportunities may be precluded by these conversion costs, which for many combinations of factors (site class, forest type, timber prices, regeneration intensity, and interest rate) result in negative LEVs. Improved technology and/or knowledge of reforestation practices in these situations may provide opportunities to reduce the costs of converting many of these sites as research continues into these practices.

Who carries the burden of grassland to forest conversion costs obviously plays a major role in determining profitability of conversion for a landowner, according to our results. The question of who should carry this cost burden, based on the tenets of SMCRA mandating that mined land be reclaimed in a fashion that restores the land to a condition capable of supporting pre-mine uses, or higher and better uses, is an important question that continues to beg for an answer. For previously reclaimed lands, the question carries with it two issues: (1) the legal question of who is responsible under the law when performance bond release already has been granted, and (2) the technical question of what constitutes a condition capable of supporting pre-mine (or higher) uses. Presumably only the latter issue is relevant for future reclamation, but its answer should involve the regulatory agencies that must recognize and judge the adequacy of reclamation efforts, and particularly reforestation. These agencies play an important role in this process, as they must understand what is best from the landowner's perspective. Mine operators may have incentives to opt for lowest cost options, while we see in many cases that medium to high regeneration intensity is best for the landowner, and may be necessary to truly meet the condition that the land support pre-mine or higher uses.

It also appears that in many cases substantial payments, non-revenue values, or carbon values are required to reach “profitability” under the present circumstances. It is unclear when, or in what form, markets will develop to support any of these add-on values to supplement commercial forestry revenues. However, as these markets do develop, they will only enhance the viability of forestry on reclaimed mined lands, although as we demonstrate in our analysis of carbon payments, the form of the revenue source may itself influence management, potentially mitigating some of the benefits of reforestation.

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Appendix A: Glossary of Symbols

Symbol	Definition
$V(.)$	landowner indirect utility function
R	total revenue generated from previously reclaimed surface-mined property
I	exogenous landowner income
L_0	condition of the property under current management
L_f	condition of the land after reforestation
Φ	amenity benefits derived from owning the land
i	interest rate
$a(.)$	annual revenues generated from the land (e.g., grazing or hunting leases)
$b(.)$	amenity benefits derived from owning land each year
z	a variable of integration representing time periods $1, \dots, \infty$
p	unit price of timber
$Q_f(t)$	volume of timber produced from reforested land at age t (rotation length)
c_f	regular reforestation costs after timber harvest
c_0	initial reforestation costs incurred when converting grassland into forest
$x(z)$	quantity of carbon accumulated on the forested site at time z
$\dot{x}(z)$	annual carbon increment in year z , or $\dot{x}(z) = dx/dt$
s_{ca}	payment per ton of accumulated carbon found on site each year
s_{cg}	payment per ton of incremental carbon grown on the site that year

TASK 5: Determine the potential of large-scale post-SMCRA (Surface Mining Control and Reclamation Act) grassland reforestation to sequester atmospheric carbon.

Executive Summary

Many reclaimed coal-surface mine areas in the eastern U.S. are not in productive use. Reforestation of these lands could provide societal benefits, including environmental services such as watershed protection, sequestration of atmospheric carbon (C), and production of wood products. Currently, little information is available on the characteristics and extent of the land resource base created under SMCRA. Our objectives were to determine the suitability of eastern US coal-mined lands' soil properties for reforestation and to estimate the cumulative C sequestration potential of such lands if they were to be successfully reforested

Electronic databases of coal mining permits issued under SMCRA were obtained from SMCRA-permitting agencies in the states of KY, OH, VA, and WV. The time periods covered by the databases varied, but all extended back until at least the early 1990s. Twenty bond-released permits were selected from each state's database using an area-weighted randomization procedure, and a variety of means were employed in an effort to identify current landowners and obtain permission to access the mined properties for sampling. In some cases, substitutes for selected sites were accessed, when knowledgeable parties affirmed that a substitute was both similar in character to a selected but inaccessible property and accessible for sampling. Six mine sites each in OH, KY, and VA and seven sites in WV were sampled at up to 10 randomly selected points per site during the summer and early fall of 2005. The 25 sites yielded 225 sample locations. At each sampling point, soil physical properties (including bulk density, coarse fragment content, particle-size distributions) were determined for the top 30 cm; soil chemical properties (including pH, exchangeable base cations, extractable P, electrical conductivity, soil C, and soil nitrogen) were determined for soil samples split at the topsoil – subsoil interface (0-10 cm, and 10-30 cm); and site attributes (including vegetative community type; herbaceous biomass accumulation; and hand-held auger penetration depth) were determined for each sampling location. Measured soil properties were used to estimate forest site productivity, expressed as white-oak site index at age 50 (SI50) at each sampling location using two methods: A method developed by Jones (2005) through the current project (“Jones Model”), and method developed by the investigators based on past research findings of J.A. Burger (“General Model”). SI50s estimated using the Jones Model method varied from 19.5 to 30.0 m, and General-Model-estimated SI50s varied from 22.1 to 26.3 m. No significant differences in among state-mean SI50s were found to occur using either SI50 estimating method. The mean SI50s for all sites sampled were similar for the two estimating methods: 25.1 for the General Model and 25.5 for the Jones Model.

In order to estimate the potential of lands mined for coal under SMCRA to sequester atmospheric C, if those lands were to be successfully reforested, those lands' total acreages were estimated from US Office of Surface Mining data, assuming 50% of the total of Phase III bond released lands (1978-2005) and 50% of Phase I bond-released lands over the 2001-2005 period in the study region (eastern OH, PA, eastern KY, WV, VA, MD, TN) (a total of about 300,000 hectares) to be available for reforestation. The total potential of these lands to accumulate atmospheric C was estimated, as a function of SI50, using relationships developed by Amichev (2007) through analysis of data collected by Rodrigue (2001). Assuming the sampled mine sites to be representative of the mined-land resource base, reforestation of mined lands with mixed

pine-hardwood species would result in an average estimated C accumulation in forms that can be harvested for use as wood products or are likely to remain in the soil C pool at $\sim 250 \text{ Mg C ha}^{-1}$ over the 60-year period following reforestation.

Only when C accumulates as a response to management actions in quantities over and above that which would accumulate in the absence of such management can that C be considered as “sequestered” a condition commonly termed as “additionality.” The “additionality” of this potential C sequestration was estimated considering data in scientific literature that defines C accumulation in mined-land grasslands over the long term. Given assumptions detailed in the text, these lands have the potential to sequester $\sim 180 \text{ Mg C ha}^{-1}$, a total of $53.5 \times 10^6 \text{ Mg C}$, over 60 years, an average of $\sim 900,000 \text{ Mg C / yr}$, an amount equivalent to about 0.04% of projected U.S. C emissions at the midpoint of a 60-year period (circa 2040) following assumed reforestation.

Although potential sequestration quantities are not great relative to potential national needs should an energy-related C-emissions offset requirement be developed at some future date, these lands are available and unused for other economically valued purposes and many possess soil and site properties that are well-suited to reforestation. Should such reforestation occur, it would also produce ancillary benefits by providing environmental services, such as enhanced watershed protection, and producing timber and renewable-fuel products. As time passes, these lands’ collective economic potential for reforestation can be expected to diminish due to the influence of continued proliferation and growth of low-value woody species with minimal C sequestration potentials.

Experimental

Identify and Sample Mine Sites

Twenty mine permits each in Kentucky, Ohio, Virginia, and West Virginia were selected from permitting databases maintained by state agencies using a randomized procedure where individual permits were weighted by acreage. An effort was made to identify and contact landowners so as to obtain site access permissions. For sites where access permission could not be obtained but knowledgeable parties communicated opportunities to access nearby sites with similar characteristics, these sites were sampled as substitutes for the original sites. Sampling locations were located on each site using a randomization procedure.

Determine Soil and Site Properties at Sample Locations

Field parameters measured at each sampling location include slope, aspect, depth of penetration by a hand-operated for mine-soil application using a screw auger to penetrate the soil until refusal (this measure is hereafter termed as “auger depth”) (Jones et al. 2005). Herbaceous biomass was sampled 1 x 1/4 m quadrats (3 per sampling site) and oven dry weights were obtained. Vegetation characteristics within a 20m radius of the sampling site were estimated as proportionate coverages represented bare ground, open water, herbaceous species, shrub species, and forest species; dominant species, species mixes, and/or species groups within each vegetative component were also recorded; where dominant communities varied within the 20-m radius, the area was sketched and the dominant species, mixes, and/or groups within the each sub-area were recorded. Estimated percent compositions recorded for each sampling location were averaged by mine site.

Soils were sampled by excavation of an approximate 30 x 30 x 30 cm volume, separated at 10-cm depth into surface and subsurface fractions. The volume of the excavated cavity was determined by filling the hole with small spheres and determining volume occupied by the spheres upon retrieval. Soil samples were bagged and weighed, and a subsample of the soil-sized fraction was removed to determine moisture content. After air drying, soil samples were sieved at 2 mm and the coarse fragment content (> 2 mm) was determined as a mass proportion of the total sample. Bulk density of the soil-sized fraction was determined as the dry-mass-to-volume ratio, assuming a coarse-fragment density of 2.4 g/cm^3 . Particle size distribution of the soil-sized fraction (% sand, % silt, % clay of the soil-sized fraction) was determined using the hydrometer method (Gee and Bauder 1986). Percent composition of coarse fragments within each of the following classifications was estimated visually: shale, gray siltstone, brown siltstone, unweathered sandstone, weathered sandstone, coal. Bulk density was determined on a whole-soil basis (0-30 cm), while coarse fragment content and composition were determined separately for 0-10 and 10-30 cm depths. Particle size analysis was conducted on a composite sample (1/3 surface + 2/3 subsurface).

All soil chemical properties were determined separately for the surface and subsurface soil fractions. Soil nutrients (P, K, Ca, Mg, Zn, Mn, Cu, Fe, and B) were extracted with a Mehlich 1 solution using a 1:5 vol:vol soil-to-extractant ratio and analyzed using a Thermo Elemental ICAP 61E (Inductively Coupled Argon Plasma Atomic Emission Simultaneous Spectrometer) by Virginia Tech Soil Testing Laboratory. Soil pH was determined on a mixed suspension of 1:1 volume-to-volume ratio of soil material to distilled water (McLean, 1986), buffer pH was determined by using a similar procedure but replacing distilled water with a Mehlich 1 buffer solution, and soluble salts using the method of Rhoades (1986) by measuring the electrical conductivity of a 1:2 volume:volume ratio of soil material to distilled water, also by the Virginia Tech Soil Testing Laboratory.

Sodium-bicarbonate extractable P was determined using the method of Olsen and Summers (1986), and exchangeable cations (Ca, Mg, K, Na) were extracted with 1 M ammonium acetate (Thomas, 1986). Phosphorus and cations were determined using a Thermo Elemental ICAP 61E (Inductively Coupled Argon Plasma Atomic Emission Simultaneous Spectrometer) by Virginia Tech Soil Testing Laboratory. Exchangeable acidity was determined using the potassium chloride extraction technique (Thomas, 1986). Effective cation exchange capacity (ECEC) was estimated by summing the charge associated with exchangeable acidity and exchangeable Ca, Mg, K, and Na. Base saturation was calculated as the proportion of the ECEC occupied by base cations (Thomas, 1986). Total C and N were determined using a Leco C / N analyzer (LECO Corp, St. Joseph, MI).

Reclamation dates for each sampling site were estimated by applying best professional judgment while considering known information such as mine permit application and/or approval dates; performance-bond release dates (when available), recollections of contacts familiar with the sites; ages of planted eastern white pines (if present), and normalized difference vegetation indices (NDVI, a spectral indicator of vegetation density) derived from Landsat satellite imagery dated 1987, 1988, 1994, 1998, and 2000 for the Virginia and southern West Virginia sites. A median reclamation age (calculated as 2005 minus estimated year of reclamation) was calculated for each site.

Estimate Forest Productivities

The forest productivity of each sampling site was estimated as 50-year site index for white oak (expressed in feet) using a method developed by Jones et al. (2005), as a component of the current project, and through development of a generalized model, which was validated using field data collected by Rodrigue (2001).

Estimate Forest Productivities: Jones Model

Forest productivity “sufficiency curves” developed by Jones et al. (2005) were used to determine sufficiency levels for each soil measurement at each sampling location and a forest site productivity index was determined for each sampling location, also using Jones et al. (2005). Sufficiency values are assigned to soil parameters at each sampling location on a 0 – 1 scale, with 0.0 indicating poor forest site quality and 1.0 indicating superior quality (See Appendix A). These values are combined to calculate a productivity index using the following equation:

$$PI_{JONES} = 0.44 * S_{BD} + 0.28 * S_{AD} + 0.20 S_{s+c} + 0.08 S_{PH} \quad (1)$$

where:

PI_{JONES} = Productivity Index (Jones method).

S_{BD} = Bulk density of the soil-sized fraction (g/cm^3) sufficiency.

S_{AD} = Auger depth (cm) sufficiency.

S_{s+c} = Soil texture (% of soil-sized fraction comprised of silt + clay) sufficiency.

S_{PH} = Soil pH sufficiency.

Jones *et al.* (2005) developed and validated this method using samples taken from the top 20 cm. Our sampling depths were 30 cm. For properties analyzed for the full 30 cm depth (texture, bulk density), measured values were used to determine sufficiency levels. For properties analyzed separately for 0-10 cm and 10-30 cm layers (pH), a “whole soil” value was calculated by considering component values as proportionate to sampling depth.

A mean PI value was calculated for each mine site, and state and overall means were calculated from the mine-site means. The productivity index for each sampling site was converted to a 50-year site index for white pine, expressed in feet ($SI50_{WP}$), using an equation derived by Jones et al. (2005), Figure 7:

$$SI50_{WP} = 117.74 * PI_{JONES} + 15.588 \quad (2)$$

An equivalent 50-year white oak site index expressed in feet ($SI50_{WO}$) was calculated for each sampling location, using a relationship derived from Figure 1 in Doolittle (1958):

$$SI50_{WO} = 0.8627 * SI50_{WP} - 5.7647 \quad (3)$$

Estimate Forest Productivities: General Model

Because the Jones method for estimating forest productivity relies primarily on physical parameters, utilizes a measure that proved difficult to apply with consistent results in the field (auger depth), and was validated using measurements from only a single species (eastern White Pine), the investigators developed an alternative method for estimating soil productivity, termed hereafter as the General Model.

Based on his research experience (including Andrews et al., 1992; Jones et al. 2005; Showalter et al. 2006; Torbert et al. 1988; Torbert et al. 1992; and other studies), the principal investigator (J.A. Burger) developed an expression for calculating a qualitative productivity

index:

$$PI_{GEN} = (S_{BD} + S_{CF} + S_{TX} + S_{BS} + S_{PH} + S_{EC} + S_P) / 7 \quad (4)$$

where:

PI_{GEN} = Productivity Index (General model)

S_{BD} = Bulk density of the soil-sized fraction (g/cm^3) sufficiency.

S_{CF} = Coarse fragments (% of soil mass) sufficiency

S_{TX} = Soil texture (% of soil-sized fraction comprised of silt + clay) sufficiency

S_{BS} = Base saturation (%) sufficiency

S_{PH} = Soil pH sufficiency

S_{EC} = Electrical conductivity (dS/m, 1:5 soil:water equivalent) sufficiency.

S_P = Bicarbonate-extractable phosphorous (mg/kg) sufficiency

Sufficiency curves used to construct the General Model are listed in Appendix B. The electrical conductivity values used to develop the General Model were based on a 1:5 soil:water EC analyses, while the current study measured EC using a 1:2 soil:water ratio; therefore, an additional EC measurement was obtained 32 of the collected samples, selected from the sample set using a randomized procedure biased towards higher values so as to represent the full range of EC measurements, using the 1:5 ratio test (Rhoades 1986); these data were used to generate a regression equation which was applied to estimate equivalent 1:5 soil:water-ratio EC values for each sample, based on the 1:2 soil:water-ratio EC measurement (Fig. 5-1).

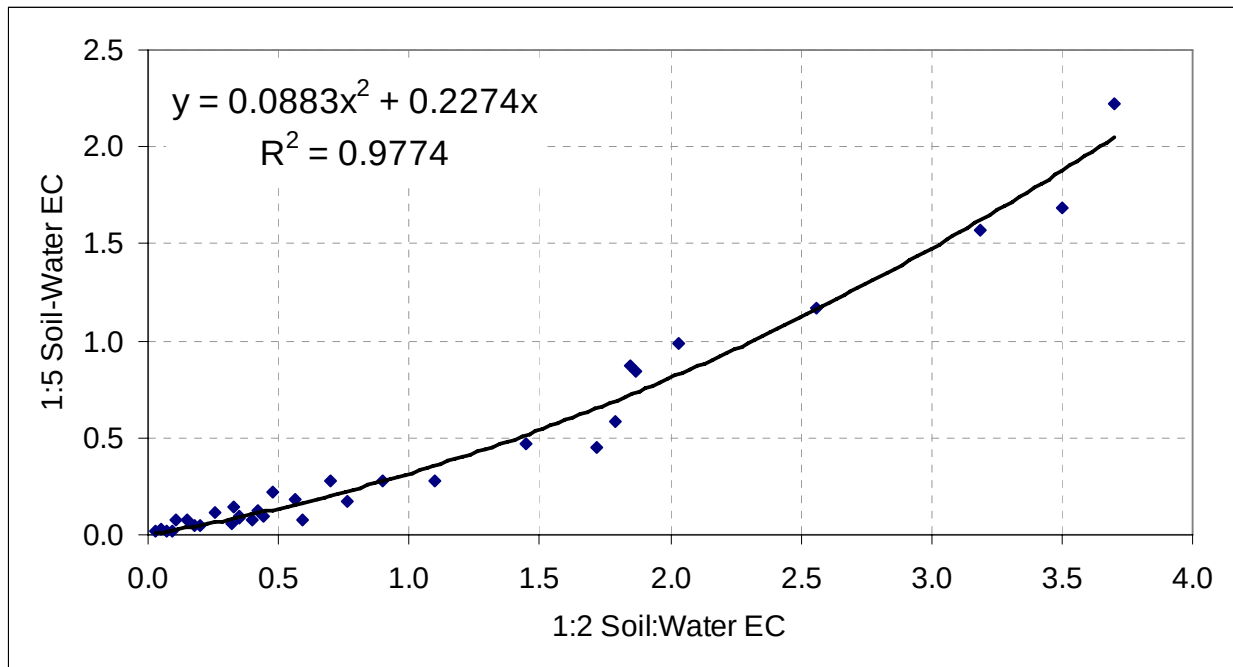


Figure 5-1. Correspondence of 1:5 soil:water EC and 1:2 soil:water EC for 32 soil samples selected to represent a range of measured 1:2 EC values, and the equation that was used to estimate equivalent 1:5 soil:water EC values based on measured 1:2 soil:water EC.

The adequacy of the PI_{GEN} as an indicator of site index was validated using data collected by Rodrigue (2001) in his study of forest site productivity in mined land forest sites in eastern USA. Using Rodrigue's data (mined sites only, excluding non-mined control sites; 3 to 4 sampling locations per site, at 11 sites in IL, KY, OH, PA, and WV, with the VA site excluded from analysis because data were incomplete and the IN sites excluded because of uneven aged stand characteristics), sufficiency values and a PI_{GEN} value were calculated for each sampling location, and a mean PI_{GEN} was calculated for each site. Rodrigue had measured site index (SI) for the dominant species at each site, and converted that measured SI value to a 50-year white-oak site index (SI_{50WO}). PI_{GEN} values were regressed against SI_{50WO} to test the validity of PI_{GEN} as a site-index indicator, and to define an SI_{50} predictive relationship.

The forest productivity of each of the current study's sampling sites was estimated by applying the General Model method to whole-soil values. For properties analyzed over the full 30 cm sampling depth (texture, bulk density), measured values were used as estimates of "whole soil" values and to determine sufficiency levels. For properties analyzed separately for 0-10 cm 10-30 cm layers (pH), a "whole soil" value was calculated assuming a depth of 78 cm (as per Andrews et al. 1992); the 0-10 cm value was considered as representative of 0–10 cm layer, the 10-30 cm value was considered as representative of soil properties extending from 10 – 78 cm, and a "whole soil" value was calculated by considering component values as proportionate to representative depth (i.e., with the 0-10 cm measurement contributing 12.8% to the whole-soil value). Sufficiency levels were determined using the whole-soil values.

Estimate C Sequestration Potentials of Sampled Mine Sites:

The potential for each mine site to accumulate C was estimated using equations developed by Amichev (2007). Amichev (2007) estimated the potentials of various stand types on mined lands, based on data collected by Rodrigue (2001).

$$\text{For pines: } Eco\_C = \text{Exp}(4.416 + 0.779 * SI / \text{age}) \quad (5)$$

$$\text{For mixed stands: } Eco\_C = \text{Exp}(4.147 + 1.039 * SI / \text{age}) \quad (6)$$

$$\text{For hardwoods: } Eco\_C = 219.455 - 3711.777 * (1 / \text{age}) \quad (7)$$

where:

Eco_C = C accumulation as above- and below-ground biomass, litter, and soil organic materials on reforested mine sites (Mg C/ ha).

SI = 50-year white-oak site index, expressed in meters.

Age = Average age, in years, of the dominant and co-dominant trees.

Si estimates were converted to C sequestration-potential estimates for each stand type on each mine site, considering:

- C quantities available for conversion into atmospheric-carbon-neutral uses (wood and fuel products, which are assumed to substitute for fossil fuels),
- C quantities likely to be maintained in the forest system through a harvest and regeneration; and
- "additionality," the C-accumulation increment expected to occur as a response to

reforestation (i.e., C accumulation over and above that which would occur in the absence of reforestation), assuming the mine sites to remain in grass vegetation if not actively reforested.

Estimate C Sequestration Potentials and Significance of Post-SMCRA Mine Lands:

The total area of land disturbed and reclaimed by mining operations in eastern US states under SMCRA was estimated from SMCRA bond-release data obtained from US OSMRE.

The distribution of soil and site conditions, and forest productivities, on the sampled mine sites were considered to be representative of the distribution of soil and site conditions on post-SMCRA mined areas. Considering these data, information gained through consultation with state and federal mining agency officials, the investigators' experience, and best professional judgment, the proportion of reclaimed mined areas potentially "available" for reforestation was estimated and applied to the SMCRA bond-release acreage totals to estimate the total acreage of post-SMCRA mined and reclaimed lands potentially "available" for reforestation.

The potential significance of these lands' C sequestration potentials was estimated assuming lands would be reforested over a multiple-year period centered on the year 2010. Since these lands' C sequestration potentials were estimated for a 60 year period, the mid-point of that 60-year period (2040) was selected as a reference for use in determining the mined lands' potential significance. Total US energy-related C (as CO₂) emissions are projected by US Energy Information Administration (EIA 2006) for the year 2030 ("Reference Case" forecast, Table A2); a 2040 "business as usual" US emissions estimate was developed by assuming the 1.2% average annual increase projected for the 2004-2030 period by US EIA (2006) would continue forward until 2040. Other reference "benchmarks" for determining the significance of mined lands' estimated sequestration potentials were also developed, as explained in the text that follows.

Results and Discussion

Property Access

Six mine sites were sampled in each of three states (OH, KY, and VA) and seven mines were sampled in WV (Fig. 5-2). The 25 sites yielded 225 sample locations (Table 5-1).

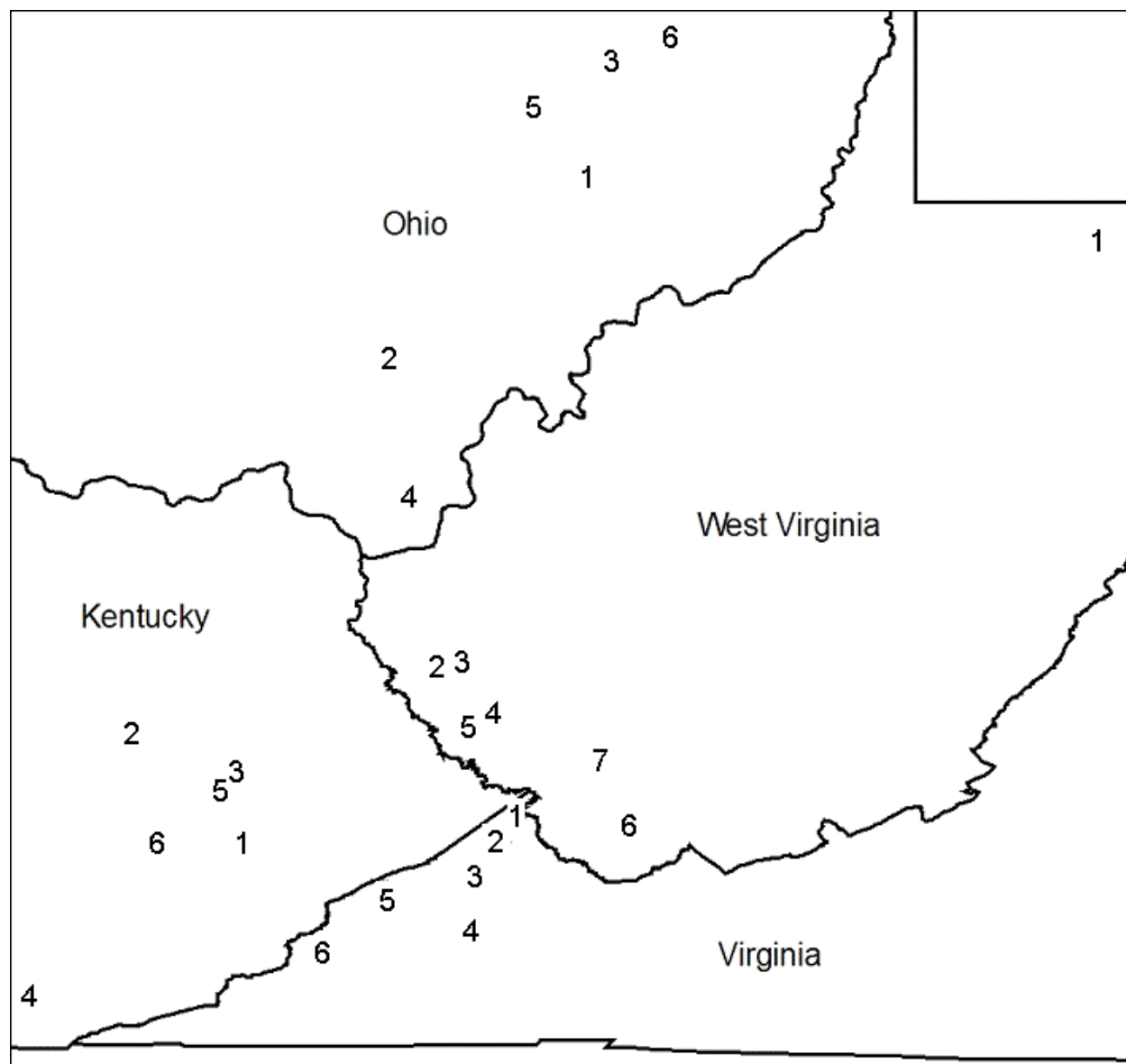


Figure 5-2. Mine site sampling locations. Numbers correspond with state-site numbers on Table 5-1.

Table 5-1. Mine sites where soils and vegetation were sampled and characterized.

Mine Site	Sampling Date	No. of Samples	County	Reclam. Date[†]	Predominant Cover/Land Use
KY-1	8/16/05	5	Breathitt	1990	Abandoned grasses
KY-2	8/17/05	10	Wolfe	1996	Abandoned grasses
KY-3	8/18/05	10	Magoffin	1996	Abandoned grasses
KY-4	8/31/05	10	Knox	1990	Mixed grasses & scrub woody (black locust)
KY-5	9/2/05	3	Breathitt	1996	Reforested with pines
KY-6	9/27/05	10	Perry	1995	Abandoned grasses
OH-1	7/19/05	10	Muskingum	1987	Abandoned grasses
OH-2	10/4/05	9	Vinton	1990	Abandoned grasses
OH-3	10/6/05	10	Coshocton	2000	Abandoned grasses
OH-4	10/18/05	10	Gallia	1985	Wildlife Habitat
OH-5	10/19/05	9	Muskingum	1986	Wildlife Habitat
OH-6	10/20/05	6	Tuscarawas	1980	Grasses (Pastured until recently)
VA-1	6/1/05	8	Buchanan	1989	Mixed grasses & scrub woody (Black locust & red maple)
VA-2	6/8/05	9	Buchanan	1986	Forest - mixed hardwood & pines
VA-3	6/15/05	10	Buchanan	1988	Forest - mixed hardwood & pines
VA-4	6/22/05	10	Russel	1998	Mixed white pine & scrub woody (black locust)
VA-5	8/9/05	10	Dickenson	1995	Grasses (Pastured until recently)
VA-6	9/26/05	10	Wise	1998	Mixed grasses & scrub woody (autumn olive)
WV-1	7/27/05	10	Preston	1990	Abandoned grassland / scrub woody
WV-2	8/3/05	9	Mingo	1985	Scrub woody (mixture of planted and invasive hardwoods, black locust, and autumn olive.
WV-3	8/4/05	10	Logan	1994	Mixed grassland / scrub (black locust & autumn olive)
WV-4	9/7/05	8	Logan	1987	Mixed grasses & black locust.
WV-5	9/8/05	9	Logan	1995	Abandoned grasses
WV-6	9/20/05	10	McDowell	1984	Mixed grasses & scrub woody (autumn olive and other)
WV-7	9/21/05	10	Wyoming	1990	Scrub woody (black locust and other)

[†] Estimated average for sampling locations, based on available information.

We selected more sites for sampling than available resources would allow to be sampled, expecting that site-access problems would occur. However, we did not fully anticipate the level of effort that would be required to obtain site access. Problems encountered are associated with identifying property owners (although the permit applications filed by mining firms with regulatory agencies contain landowner information, the mine permit databases do not, and landowner information that we were able to obtain from selected permits was outdated), contacting the owners (those ownership records we were able to obtain from agency contacts were out of date; some sites were listed as having multiple owners; some of the corporate owners identified by the permits were no longer in business, while private owners were difficult to locate), and obtaining sampling permissions from those which we were able to identify and contact (although most owners were agreeable, some required that right-of-access agreements be drafted and signed, which requires involvement by both landowner and university legal personnel, while a small minority declined to cooperate).

Twenty-five mine sites were accessed in four states, including 17 sites on the randomly selected list and 8 substitutes (Fig. 5-2, Table 5-1).

Our intention was to obtain 10 samples per site, and 10 samples were obtained from 15 of the 25 sites. Randomization of sampling site placement resulted in location of an intended sampling point in a roadway or other inconvenient location on some mine sites. In these cases, the number of sampling points was reduced to nine. There were several cases when the allotted time did not allow completion of the planned sampling activity but scheduling required that sampling be terminated due to commitments in other locations. On KY-1, access permission from a responsible party was obtained prior to the site visit (as on all other sites) but the hostility of personnel encountered at the site caused sampling to terminate after only five samples had been obtained. Sampling personnel were delayed in accessing KY-3 because of a misunderstanding about meeting place and time with the personnel who were to guide them to the site, which was more remote than expected and characterized by difficult terrain. These events combined with subsequent scheduling obligations allowed only three samples to be obtained at KY-3. OH-6 was the final site sampled; after six samples were obtained, weather conditions deteriorated with several days of poor weather predicted so sampling was terminated.

Vegetative Cover

Vegetative cover characteristics were highly variable (Tables 5-2 and 5-3). In order to compute mine site means, dominant vegetation species and communities recorded were grouped. Because of high variability and non-normal distributions, statistical comparisons were performed only on major species categories using a non-parametric procedure (Table 5-3) and, despite major differences between mean values, few statistical differences were found. The predominant planted-herbaceous species on all sites was K-31 fescue (*Schedonorus phoenix* (Scop.) Holub). Although sericea lespedeza (*Lespedeza cuneata* (Dum.-Cours.) G. Don) was present as a minor component of the vegetation mix on many sites, especially on VA, KY, and WV, it was rarely dominant or co-dominant. The highest percentage cover for planted herbaceous species was computed for OH, but no statistical differences between states were found.

Table 5-2. Mean values estimated percent cover provided by dominant vegetation types at sampling locations among mine sites, with mine site means averaged by state and for all mine sites sampled.

State - Site	<u>- Planted Herbaceous -</u>					<u>- - - Native Woody - - -</u>							
	Fescue	Other Grasses	Lespedeza	Other Herbaceous Legumes	Native Herb, Shrub, Vine	Black locust	Red maple	Other Native Hardwood	Native Softwood	Invasive	Other	Bare	Water
KY-1	64	-	-	2	14	-	-	-	-	-	-	20	-
KY-2	54	5	7	30	2	1	0	-	1	-	-	-	-
KY-3	27	3	6	47	-	-	-	-	3	3	-	11	-
KY-4	40	-	24	-	6	29	-	-	-	-	-	1	-
KY-5	-	-	-	-	-	28	-	17	47	8	-	-	-
KY-6	31	1	38	16	1	1	-	1	-	1	-	11	-
OH-1	48	-	-	3	39	-	-	8	-	1	-	-	1
OH-2	50	-	9	31	11	-	-	-	-	-	-	-	-
OH-3	36	15	-	12	37	-	-	-	-	-	-	-	-
OH-4	57	3	20	7	10	-	-	-	-	1	3	-	-
OH-5	26	27	-	22	25	0	-	-	-	-	-	-	-
OH-6	83	2	-	-	-	-	-	7	-	9	-	-	-
VA-1	7	-	1	-	26	17	23	7	-	16	3	-	-
VA-2	1	-	-	-	-	-	20	10	48	14	-	6	-
VA-3	6	-	-	-	2	1	19	17	47	8	-	-	-
VA-4	52	-	-	10	9	16	-	-	4	9	-	-	-
VA-5	24	-	70	1	3	-	-	2	-	1	-	-	-
VA-6	30	-	8	8	13	9	-	4	1	24	-	5	-
WV-1	91	-	-	-	2	3	-	-	-	2	-	2	-
WV-2	20	-	13	9	20	14	5	12	3	4	-	-	-
WV-3	45	-	8	22	7	7	-	-	3	8	-	-	-
WV-4	2	-	9	-	19	43	14	4	6	1	-	3	-
WV-5	49	-	11	8	15	0	-	-	-	3	-	13	-
WV-6	16	1	12	-	8	22	9	6	2	24	-	-	-
WV-7	14	-	1	-	2	52	-	0	-	30	-	1	-
KY	36	2	12	16	4	10	0	3	8	2	-	7	-
OH	50	8	5	12	20	0	-	2	-	2	0	-	0
VA	20	-	13	3	9	7	10	7	17	12	0	2	-
WV	34	0	8	5	10	20	4	3	2	10	-	3	-
All	35	2	9	9	11	10	4	4	7	7	0	3	0

Table 5-3. Mean values estimated percent cover provided by dominant vegetation types at sampling locations among mine sites, with mine site means averaged by state and for all mine sites sampled.

State-Site	Total Planted Herbaceous	Native		
		Herbaceous Shrub, Vine	Total Native Woody	Invasive
KY-1	66	14	-	-
KY-2	96	2	2	-
KY-3	83	-	3	3
KY-4	64	6	29	-
KY-5	-	-	92	8
KY-6	85	1	2	1
			-	
OH-1	51	39	8	1
OH-2	89	11	-	-
OH-3	63	37	-	-
OH-4	87	10	-	1
OH-5	75	25	0	-
OH-6	84	-	7	9
			-	
VA-1	9	26	46	16
VA-2	1	-	78	14
VA-3	6	2	84	8
VA-4	63	9	20	9
VA-5	95	3	2	1
VA-6	45	13	14	24
			-	
WV-1	91	2	3	2
WV-2	42	20	34	4
WV-3	75	7	11	8
WV-4	11	19	67	1
WV-5	68	15	0	3
WV-6	29	8	39	24
WV-7	15	2	52	30
			-	
KY	66 ^a	4 ^a	21 ^{ab}	2 ^{ab}
OH	75 ^a	20 ^a	3 ^b	2 ^b
VA	36 ^a	9 ^a	41 ^a	12 ^a
WV	47 ^a	10 ^a	29 ^{ab}	10 ^{ab}
			-	
All Sites	56	11	24	7

[†] State means followed by different letters are significantly different ($\alpha = .05$), as determined via Tukey's HSD test applied to ranks of site means.

Numerous species – choke berry (*Photinia sp.*), greenbrier (*Smilax sp.*), poison ivy (*Toxicodendron radicans* L.), ladies lace (*Daucus carota* L.), sumac (*Rhus sp.*), and Virginia creeper (*Parthenocissus quinquefolia* L.) – were found on single or small numbers of sites and included in the “native herbaceous, shrubs, and vines” category, which represents non-forest tree native species that were unlikely to have been planted and apparently invaded from surrounding terrain. Blackberry (*Rubus sp.*) was identified as a dominant or co-dominant shrub species at one or more sampling locations on 13 of the 25 sites. Goldenrod (*Solidago sp.*), ragweed (*Ambrosia sp.*), thistle (*Cirsium sp.*), and aster (*Aster sp.*) were the primary species recorded as dominant on the Ohio sites.

Few woody species were found on the OH mine sites. On the VA, WV, and KY mine sites, black locust (*Robinia pseudoacacia* L.) was the most common native woody species found, which is not surprising since it is widely planted. Black locust, however, does not produce high-quality timber when planted on mine sites. Red maple (*Acer rubrum* L.) was recorded commonly in WV and VA in patterns that appear to have occurred through natural invasion. Other native woody species with the potentials to form harvestable tree crops – including Virginia pine (*Pinus virginiana* P. Mill.), eastern white pine (*Pinus strobus* L.), and hardwood species such as oaks (*Quercus sp.*) and tulip poplar (*Liriodendron tulipifera* L.) – were found to occur, with the greatest nominal frequencies in VA and WV, apparently in most cases due to reclamation plantings. Native woody species were recorded as occurring more commonly as dominant and co-dominant vegetative components at VA sites than at OH sites (Table 5-3).

Autumn olive (*Elaeagnus umbellata* Thunb.) was the most common invasive species recorded, followed by multiflora rose (*Rosa multiflora* Thunb.). Autumn olive was recorded as a dominant or co-dominant vegetative component on all VA and WV sites, often at several sampling locations. Invasives were recorded as occurring more commonly as dominant and co-dominant vegetative components at VA sites than at OH sites (Table 5-3).

On sites where herbaceous biomass was sampled, amounts ranged from 506 to 1784 kg ha⁻¹, and the mean value was 1136 kg ha⁻¹ (Table 5-4). No statistical differences among states were noted. Herbaceous biomass was not collected at two Virginia sites where forest cover had closed canopy.

Soil Properties

Although soil properties varied widely among sampling sites, few significant differences were found (Tables 5-4 and 5-5). Auger penetration depth proved to be a difficult procedure to implement in a manner that yielded consistent results in the field. The procedure calls for the auger to be inserted in an effort to obtain penetration up to three times. Auger penetration depths were recorded as “refused” at 10 percent of the sampling sites, primarily due to the effects of large coarse fragments. Bulk densities were generally not excessively high. Site mean bulk densities of the soil-sized particles ranged from 0.93 to 1.54 g cm⁻³, with a mean value for all sites of 1.21 g cm⁻³, while site-mean coarse fragment contents ranged from 23 to 62% with a mean value of 50%. Silt + Clay (S+C) contents were nominally greatest in OH, at 73%, an apparent result of that state’s requirement that topsoils or soil-like topsoil substitute materials be used to construct a surface medium.

Table 5-4. Numbers of samples collected, mean values for herbaceous cover and soil physical properties at sampling locations by mine site, by mine site and state[†].

State-Site	Samples	Herbaceous cover (kg ha ⁻¹)	Auger depth (cm)	Bulk Density (g cm ⁻³)	Coarse Fragments (%)	Silt + Clay (% of soil-size frags)
KY-1	5	1303	30	0.93	58	64
KY-2	10	1653	32	1.01	60	62
KY-3	10	1016	14	1.15	57	55
KY-4	10	1251	29	1.23	55	73
KY-5	3	1115	30	1.02	58	48
KY-6	10	964	29	1.41	48	71
OH-1	10	1784	32	0.94	44	83
OH-2	9	1098	34	1.22	44	75
OH-3	10	939	36	1.37	37	64
OH-4	10	804	31	1.47	23	79
OH-5	9	931	37	1.25	38	66
OH-6	6	506	29	1.17	45	72
VA-1	8	1645	38	1.18	41	39
VA-2	9	n.d.	38	0.96	51	58
VA-3	10	n.d.	40	1.02	48	54
VA-4	10	1227	27	1.11	53	62
VA-5	10	1091	35	1.54	62	64
VA-6	10	610	24	1.22	47	77
WV-1	10	1226	30	1.53	43	46
WV-2	9	1228	22	1.14	52	59
WV-3	10	1369	24	1.08	60	68
WV-4	8	1168	23	1.12	51	40
WV-5	9	591	27	1.40	54	36
WV-6	10	1343	33	1.28	56	57
WV-7	10	1051	20	1.24	61	62
KY	48	1217 ^a	27 ^a	1.13 ^a	56 ^a	62 ^{ab}
OH	54	1010 ^a	33 ^a	1.24 ^a	38 ^b	73 ^a
VA	57	1143 ^a	34 ^a	1.17 ^a	51 ^a	59 ^{ab}
WV	66	1140 ^a	25 ^a	1.26 ^a	54 ^a	53 ^b
All Sites	225	1136	30	1.21	50	62

[†] State means followed by different letters are significantly different ($\alpha = .05$), as determined via Tukey's HSD test applied to site means.

n.d. = values not determined.

Table 5-5. Mean (all but C and N) and median (C and N) values for soil chemical properties[†] at sampling locations by mine site, and mean values of mine site means by state and total[‡].

State-Site	pH	EC (dS m ⁻¹)	CEC (meq/100 g)	BS (%)	P (mg kg ⁻¹)	C (Mg ha ⁻¹)	N (Mg ha ⁻¹)
KY-1	4.41	0.56	12.4	76	2.85	31	1.0
KY-2	5.53	0.06	7.7	88	3.19	21	0.8
KY-3	6.87	0.03	6.5	99	1.46	20	0.8
KY-4	6.04	0.03	8.6	92	1.67	19	1.1
KY-5	4.98	0.03	5.1	79	3.73	19	0.9
KY-6	5.67	0.33	12.7	92	2.14	34	1.2
OH-1	7.18	0.07	24.8	100	4.26	20	1.4
OH-2	6.48	0.04	10.9	93	2.75	18	1.4
OH-3	5.95	0.03	8.4	93	6.44	24	1.3
OH-4	7.75	0.08	26.7	100	2.17	56	2.2
OH-5	6.00	0.05	9.3	96	4.34	23	1.9
OH-6	5.58	0.05	10.5	95	5.73	32	2.4
VA-1	5.22	0.01	4.2	75	2.93	24	1.0
VA-2	4.85	0.02	6.3	66	5.24	23	1.2
VA-3	5.24	0.02	6.4	64	4.19	30	1.3
VA-4	6.34	0.07	9.1	98	3.38	17	0.8
VA-5	6.20	0.08	10.1	97	4.28	10	0.8
VA-6	6.56	0.11	10.9	99	1.50	18	1.1
WV-1	5.37	0.04	5.7	88	2.20	44	1.4
WV-2	5.38	0.03	6.9	83	2.20	27	1.3
WV-3	6.41	0.04	11.0	95	3.31	13	0.8
WV-4	4.93	0.02	4.0	69	3.87	25	1.2
WV-5	5.79	0.07	4.4	82	2.64	28	0.8
WV-6	5.31	0.02	7.3	92	3.95	32	1.3
WV-7	5.87	0.02	8.4	97	4.38	28	1.2
KY	5.58 ^a	0.17 ^a	8.8 ^a	88 ^a	2.51 ^a	24 ^a	1.0 ^b
OH	6.49 ^a	0.05 ^a	15.1 ^a	96 ^a	4.28 ^a	29 ^a	1.8 ^a
VA	5.73 ^a	0.05 ^a	7.8 ^a	83 ^a	3.59 ^a	20 ^a	1.1 ^b
WV	5.58 ^a	0.03 ^a	6.8 ^a	86 ^a	3.22 ^a	28 ^a	1.2 ^b
All Sites	5.92	0.07	9.7	88	3.35	25	1.2

[†] Soil properties expressed for top 30 cm for C and N, and on a whole soil basis, assuming 78 cm depth for other properties listed.

[‡] State means followed by different letters are significantly different ($\alpha = .05$), as determined via Tukey's HSD test applied to site means for normally distributed variables and to site ranks for non-normally distributed variables (EC, CEC, and BS).

Site mean pH values were generally in the range of 5 to 7, with only four sites (two in KY, one each in VA and WV) having mean pHs below 5 and only two sites (both in OH) having mean pHs > 7 (Table 5-5). A number of soil chemical properties were found to be correlated with one another and with Silt + Clay (Table 5-6), as pH, electrical conductivity (EC), cation exchange capacity (CEC) and base saturation (BS) exhibited positive and significant statistical relationships with one another and with S+C. This result is consistent with the observation that siltstones and shales, spoil types that give rise to finer-textured mine soil materials, tend to generate soils that are higher in salts, mineral-origin cation nutrients (Ca, Mg, and K), and pH than sandstone spoils.

Table 5-6. Spearman correlation coefficients of for mine-site averages of measured soil properties. ***Bold italics**** indicate statistical significance ($p < .05$).

	Age	Aug Depth	B.D.	C.F.	S+C	pH	E.C.	CEC	BS	P	C	N
Age	1											
Aug Depth	0.22	1										
B.D.	-0.03	-0.01	1									
C.F.	-0.27	-0.36	-0.27	1								
S.+C.	0.04	0.02	0.07	-0.23	1							
pH	-0.26	-0.24	0.30	-0.16	<i>0.67*</i>	1						
E.C.	-0.28	-0.14	0.12	0.01	<i>0.60*</i>	<i>0.45*</i>	1					
CEC	0.02	-0.03	0.06	-0.10	<i>0.92*</i>	<i>0.60*</i>	<i>0.73*</i>	1				
BS	-0.17	-0.30	0.30	-0.09	<i>0.68*</i>	<i>0.90*</i>	<i>0.48*</i>	<i>0.66*</i>	1			
P	0.22	<i>0.45*</i>	-0.17	0.00	-0.05	-0.26	-0.32	-0.09	-0.10	1		
C	<i>0.51*</i>	0.04	0.29	-0.33	-0.23	<i>-0.40*</i>	-0.09	-0.09	-0.28	-0.04	1	
N	<i>0.65*</i>	0.34	0.28	<i>-0.70*</i>	0.35	0.03	-0.07	0.26	0.12	0.27	<i>0.51*</i>	1

Measured mean EC values were generally low, relative to ECs typically found in fresh mine spoils, a likely result of the weathering that has occurred in these mine soils. One site-mean EC value was markedly higher than all others recorded (0.56 dS m^{-1} , at KY-1), where mean pH was 4.41 and two of five samples had pH's of less than 4, indicating a likelihood that continued pyrite weathering is the source of the salts and high E.C. Although the mean pH at KY-6, the site with the second-highest mean E.C., was over 5, pH's of less than 4 were recorded in association with high E.C. values at three of this mine site's 10 sampling locations, again indicating a likelihood that continuing pyrite oxidation is the source of the high E.C. On both KY-1 and KY-6, the $\text{pH} < 4$ sampling sites were also associated with lack of vegetative cover, areas recorded as bare ground (Table 5-2). Base saturation was generally high at the sampling sites, with 15 of the 25 mine site means at greater than 90%, a likely result of base cations released from the formerly fresh mine soils via soil weathering.

Bicarbonate extractable soil P was highly variable, with mine site means ranging from 1.5 to 6.4 mg/kg and values were well distributed throughout that range, but was significantly correlated only with auger depth, a relationship for which we are unable to interpret as meaningful. Bicarbonate extractable P levels for the 10-30 cm depth samples at 18 of the 225 sampling locations were recorded as being less than the analytical detection limit, equivalent to

0.78 mg/kg; these values were set at the detection-limit-equivalent value for the purpose of calculating means. More than 50% of the detection-limited soil P values occurred in association with pH>7.

Individual mine sites were characterized for soil N and soil C accumulation using median (rather than mean) values due to the occurrence of a small number of very high soil values (5 values exceeded 80 Mg/ha, with the highest value at 256 Mg/ha) which we considered to be outliers; in some cases, these values were associated with observations of apparent coal among the coarse fragments, but apparent coal was also observed in some samples that did not exhibit very high soil C levels. The C measurements are total C and therefore include any carbonate C that may have been present. However, we expect that carbonates were a minor to negligible component of the measured soil C values on most of these sites because carbonate strata are not common in the central Appalachian coalfields; because EC values indicate that these minesoils were well weathered; and because only 4 of the 25 mine sites had mean pH values > 6.5. Soil C measurements also include whatever fossilized-C may be present (i.e., coal particles, fossilized biogenic C embedded in the rock matrix) as well as pedogenic C, our primary interest. N also occurs as a component of coal and several high N values also occurred as apparent outliers, so N was handled similarly. Mine-site median soil C ranged from 10 to 56 Mg/ha, and mine-site median soil N ranged from 0.8 to 2.4 Mg/ha. Soil N occurred on Ohio sites in higher quantities than on other states' sites ($p < .05$), which may reflect the fact that Ohio requires topsoil replacement whereas use of topsoil-substitute materials (i.e., rock overburden materials from rock layers originating from well below the surface) to construct surface soils is common in the other states. Soil N, soil C, and site age were positively correlated with one another, indicating an apparent tendency for soil C and soil N to accumulate as ecosystems develop. The highest soil N and third-highest soil C value occurred on the oldest site (OH-6), which had been used for pasture until recently, indicating a possibility that fertility maintenance on this site had influenced these results. However, removal of OH-6 from the correlations and statistical comparisons did not alter the statistical significance of the soil N, soil C, and site age correlations.

Estimating Forest Productivity: Jones Model

Sufficiency values and productivity indices were estimated for each sampling site; sampling site productivity indices were calculated using equation (2), and averaged and transformed to SI50_{wp} values (50 year site index for white pine) by mine site. Auger depth and S+C were interpreted to be the soil properties most limiting to forest productivity (with all-mine-site mean values of 0.66 and 0.67, respectively), according to the Jones model (Table 5-7). However, bulk density and pH sufficiency values were correlated with PI_{JONES} ($p < 0.05$), while correlations of AD and S+C, although positive, were not significant at $p < .05$ (Table 5-8). A contributing factor to the significant correlation of BD with PI_{JONES} is the fact that BD is the most highly-weighted component of the Jones model. In contrast, pH sufficiency receives the lowest weighting. S+C and pH sufficiencies were correlated with one another, a finding that is consistent with the similar relationship that was found between the corresponding soil properties (Table 5-6).

Table 5-7. Mine site mean values of sufficiency values (S), the productivity index used to estimate site index using the Jones model (PI_{JONES}), and estimated 50 year site-index values calculated using the Jones Model (SI50_{WP}) with mine site means averaged by state and for all mine sites sampled.

State/ Site	S _{BD}	S _{AD}	S _{S+C}	S _{pH}	PI _{JONES}	SI50 _{WP} (m)
KY-1	1.00	0.62	0.64	0.57	0.787	33.0
KY-2	1.00	0.73	0.66	0.89	0.848	35.2
KY-3	0.85	0.34	0.77	0.63	0.673	28.9
KY-4	0.79	0.66	0.50	0.78	0.695	29.7
KY-5	0.96	0.70	0.87	0.96	0.872	36.0
KY-6	0.73	0.69	0.53	0.65	0.675	29.0
OH-1	1.00	0.73	0.35	0.54	0.758	32.0
OH-2	0.83	0.75	0.48	0.65	0.724	30.7
OH-3	0.71	0.77	0.64	0.84	0.724	30.8
OH-4	0.58	0.72	0.42	0.32	0.567	25.1
OH-5	0.90	0.77	0.62	0.87	0.807	33.7
OH-6	0.94	0.69	0.52	0.98	0.790	33.1
VA-1	0.92	0.77	0.96	0.98	0.892	36.8
VA-2	1.00	0.77	0.73	0.96	0.878	36.3
VA-3	0.98	0.73	0.78	0.92	0.865	35.8
VA-4	0.94	0.55	0.67	0.81	0.768	32.3
VA-5	0.34	0.75	0.64	0.82	0.554	24.6
VA-6	0.85	0.56	0.44	0.69	0.675	29.0
WV-1	0.44	0.70	0.91	0.98	0.652	28.1
WV-2	0.94	0.56	0.71	0.91	0.788	33.1
WV-3	0.89	0.60	0.58	0.70	0.733	31.1
WV-4	0.96	0.53	0.88	0.99	0.825	34.4
WV-5	0.62	0.63	0.93	0.71	0.692	29.6
WV-6	0.79	0.73	0.74	0.99	0.781	32.8
WV-7	0.84	0.44	0.67	0.89	0.697	29.8
KY	0.89	0.62	0.66	0.75	0.758	32.0
OH	0.83	0.74	0.51	0.70	0.728	30.9
VA	0.84	0.69	0.70	0.86	0.772	32.5
WV	0.78	0.60	0.78	0.88	0.738	31.3
All Sites	0.83	0.66	0.67	0.80	0.749	31.6

Table 5-8. Spearman correlation coefficients of sufficiency values and productivity index for the Jones model. Bold italics indicate statistical significance ($p < .05$).

	S_{BD}	S_{AD}	S_{S+C}	S_{pH}	PI_{JONES}
S_{BD}	1				
S_{AD}	0.03	1			
S_{S+C}	0.10	0.05	1		
S_{pH}	0.13	0.20	<i>0.65*</i>	1	
PI_{JONES}	<i>0.78*</i>	0.29	0.37	<i>0.57*</i>	1

Validating and Calibrating the General Model

Soil properties measured by Rodrigue (2001) on older mine sites were translated into sufficiency values and used to calculate PI_{GEN} values for each sampling site, and sampling site values were averaged to calculate a PI_{GEN} value for each mine site. Rodrigue had also estimated $SI50_{WO}$ (50-year site index for white oak) values for each site, based on measured growth rates of the dominant species.

The linear relationship between PI_{GEN} , estimated from Rodrigue's (2001) soil data, his $SI50_{WO}$ values was found to be positive and statistically significant ($p < .05$) using Spearman correlation. An equation for use in predicting $SI50_{WO}$ as a function of PI_{GEN} was generated using .jmp software (SAS Institute, Cary NC). Both the form of the $SI50_{WO}$ vs. PI_{GEN} relationship and a biological interpretation of the relationship (as soil properties continue to improve, tree growth can be expected to level off as it reaches limits due to tree genetics) suggested a functional form that approaches a horizontal asymptote. The $SI50$ prediction equation was developed using a 4-parameter logistic function:

$$Y = \Theta_1 + (\Theta_2 - \Theta_1) / (1 + \exp(\Theta_3 * (x - \Theta_4))) \quad (8)$$

where Θ_1 = function minimum value, Θ_2 = function maximum value, Θ_3 = negative of slope at inflection point, and Θ_4 = x value at inflection point. The equation was solved using the non-linear modeling component of the .jmp software by setting Θ_1 equal to zero, a level that corresponds with the minimum possible $SI50_{WO}$ value, and optimizing remaining parameters by minimizing residuals' sum-of-squares; parameter estimates generated by the software were $\Theta_2 = 86.726242851$, $\Theta_3 = -23.62200379$, and $\Theta_4 = 0.6976016252$ (Figure 3).

Estimating Forest Productivity: General Model

Sufficiency values for each component of the General Model were estimated, summed, and used to calculate PI_{GEN} for each sampling location using equation 4; means for each sufficiency value and PI_{GEN} were calculated for each mine site, and mean mine-site values were calculated for each state and for all sites sampled (Table 5-9). Using the General model, bicarbonate-extractable P was found to be the soil property that is most limiting to forest productivity ($S_p = 0.63$), while EC and BS were found to be least limiting, with mean sufficiencies of 0.97 and 0.99 respectively. Soil P and BS are indicators of soil nutrient levels. When considered together, these two factors indicate that lack of P, not the base cations (Ca^{++} , Mg^{++} , and K^+), tends to create soil fertility limitations to forest growth over time. Mean sufficiencies for the other four General Model parameters ranged from 0.80 to 0.88.

Table 5-9. Mine site mean values of sufficiency values (S) and the productivity index used to estimate site index using the general model (PI_{GEN}), with mine site means averaged by state and for all mine sites sampled.

State- Site	S_{BD}	S_{CF}	S_{S+C}	S_{pH}	S_{EC}	S_{BS}	S_p	PI_{GEN}
KY-1	1.00	0.65	0.93	0.55	0.70	1.00	0.62	0.78
KY-2	1.00	0.65	0.91	0.90	0.98	1.00	0.62	0.87
KY-3	0.85	0.75	0.95	0.83	1.00	1.00	0.39	0.82
KY-4	0.79	0.68	0.76	0.86	1.00	1.00	0.44	0.79
KY-5	0.96	0.73	1.00	0.90	1.00	1.00	0.68	0.90
KY-6	0.73	0.83	0.83	0.74	0.76	1.00	0.48	0.77
OH-1	1.00	0.93	0.65	0.75	0.98	1.00	0.74	0.87
OH-2	0.83	0.91	0.75	0.80	1.00	1.00	0.58	0.84
OH-3	0.71	0.97	0.91	0.97	1.00	1.00	0.88	0.92
OH-4	0.58	0.88	0.71	0.60	0.98	1.00	0.51	0.75
OH-5	0.90	0.93	0.89	0.96	0.99	1.00	0.75	0.92
OH-6	0.94	0.92	0.80	0.94	0.99	1.00	0.85	0.92
VA-1	0.92	0.99	0.99	0.98	1.00	1.00	0.62	0.93
VA-2	1.00	0.79	0.97	0.88	1.00	0.98	0.80	0.92
VA-3	0.98	0.78	0.97	0.87	1.00	0.83	0.68	0.87
VA-4	0.94	0.75	0.93	0.95	0.96	1.00	0.65	0.88
VA-5	0.34	0.50	0.91	0.94	0.97	1.00	0.74	0.77
VA-6	0.85	0.78	0.72	0.86	0.94	1.00	0.39	0.79
WV-1	0.44	0.88	1.00	0.98	1.00	1.00	0.53	0.83
WV-2	0.94	0.85	0.93	0.91	1.00	0.97	0.48	0.87
WV-3	0.89	0.62	0.83	0.88	1.00	1.00	0.63	0.84
WV-4	0.96	0.87	0.92	0.92	1.00	0.98	0.69	0.91
WV-5	0.62	0.79	0.92	0.82	0.97	0.93	0.53	0.79
WV-6	0.79	0.78	0.93	0.99	1.00	1.00	0.71	0.89
WV-7	0.83	0.64	0.95	0.99	1.00	1.00	0.75	0.88
KY	0.89	0.72	0.89	0.80	0.91	1.00	0.54	0.82
OH	0.83	0.92	0.79	0.84	0.99	1.00	0.72	0.87
VA	0.84	0.77	0.91	0.91	0.98	0.97	0.65	0.86
WV	0.78	0.78	0.92	0.93	1.00	0.98	0.62	0.86
All	0.83	0.80	0.88	0.87	0.97	0.99	0.63	0.85

EC, S+C, and pH sufficiencies were found to be correlated with one another, a finding that reflects similar relationships among the corresponding soil properties (Table 5-10). Bicarbonate-extractable soil P and pH sufficiencies were also found to be correlated, a finding that is not reflected by corresponding relationships among soil properties but is indicative of the well-known but non-linear dependence of P availability on pH in natural soils.

Table 5-10. Spearman correlation coefficients of sufficiency values and productivity index for the General Model. *Bold italics indicate statistical significance ($p < .05$).**

	S_{BD}	S_{CF}	S_{S+C}	S_{pH}	S_{EC}	S_{BS}	S_P	PI_{GEN}
S _{BD}	1							
S _{CF}	0.03	1						
S _{S+C}	0.18	0.13	1					
S _{pH}	0.17	0.04	<i>0.46*</i>	1				
S _{EC}	0.16	0.09	<i>0.58*</i>	<i>0.53*</i>	1			
S _{BS}	0.26	0.09	0.32	0.11	0.35	1		
S _P	0.27	0.14	0.10	<i>0.50*</i>	0.26	0.02	1	
PI _{GEN}	<i>0.40*</i>	<i>0.41*</i>	<i>0.39*</i>	<i>0.65*</i>	<i>0.61*</i>	0.16	<i>0.67*</i>	1

Six of the seven contributing sufficiencies were found to be correlated with PI_{GEN} ($p < .05$), with base saturation being the only exception. Bicarbonate-extractable P and pH sufficiencies were strongly correlated with PI_{GEN}, as was EC sufficiency ($p < .01$). The EC relationship, although statistically significant, appears to be of little consequence because 23 of the 25 sites have mean EC sufficiencies of 0.94 or greater. The soil P/pH sufficiency relationship may be indicative of on-the-ground conditions, however, given the tendency of mine spoils to bind P in insoluble forms. Bulk density, coarse fragment, and S+C sufficiencies were found to be positively correlated with PI_{GEN} ($p < .05$).

Comparing Forest Productivities Estimated Using Two Models

Mine-site mean SI50_{WO} values estimated by applying the General Model's calibration equation (Fig. 5-3) to calculated PI_{GEN} values from ranged from 22.1 to 26.2 m, while Jones-model estimated SI50_{WO} values range from 19.9 to 30.0 m (Table 5-11). Mean values for SI50_{WO} estimated using the two models, however, are quite similar (25.1 m for the General Model, 25.5 m for the Jones Model). Productivity indices (Fig. 5-4, above) and the SI50WOs generated (Fig. 5-4, below) generated by the two models exhibit highly-significant statistical relationships ($p < .001$) with one another. The significant statistical relationships are more evident visually for the productivity indices, however (Figure 5-4, below), because the Jones Model translates PI values to SI50s using a linear function while the General Model uses a non-linear transformation to estimate SI50_{WO} values.

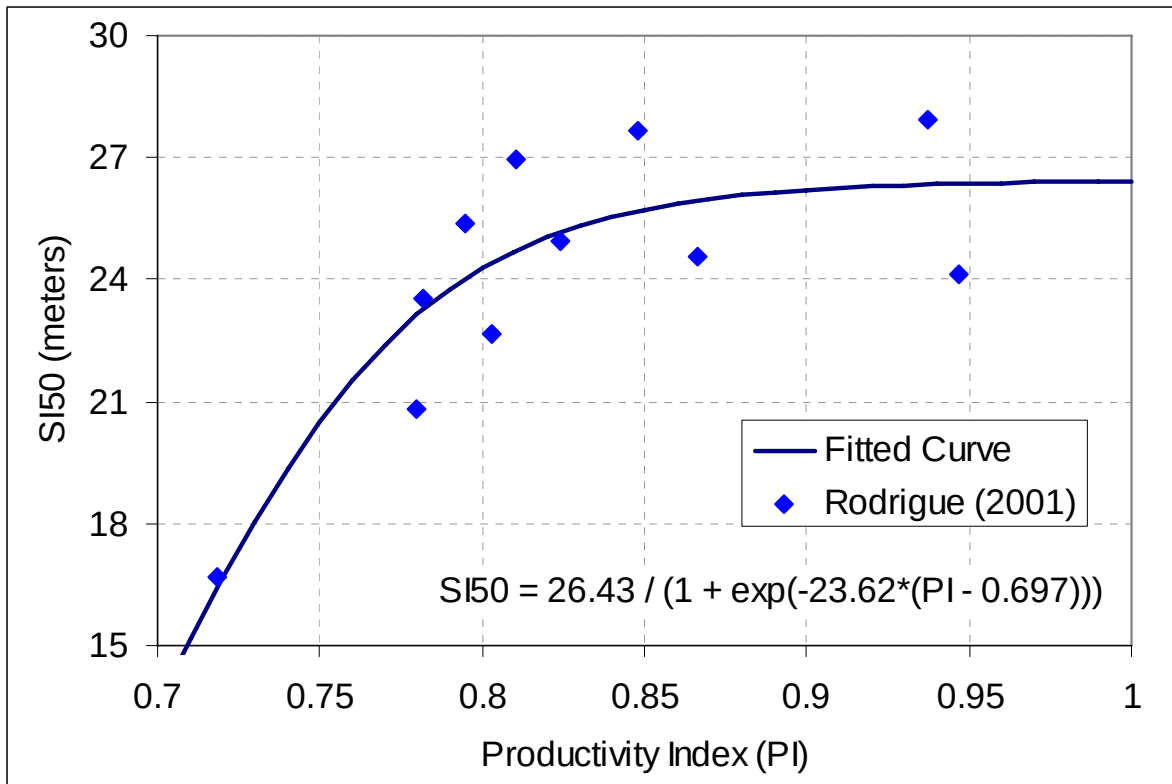


Figure 5-3. Data used for validation of the General Model.

Table 5-11. Mean values for white oak 50-year site indices (meters) estimated using the Jones Model and the General Model by mine site, with mine site means[†] averaged by state and for all mine sites sampled.

State-Site	General Model	Jones Model
KY-1	23.0	26.7
KY-2	25.9	28.6
KY-3	25.2	23.2
KY-4	23.7	23.9
KY-5	26.2	29.3
KY-6	22.1	23.3
OH-1	25.9	25.8
OH-2	25.5	24.8
OH-3	26.3	24.8
OH-4	20.6	19.9
OH-5	26.3	27.3
OH-6	26.3	26.8
VA-1	26.3	30.0
VA-2	26.3	29.5
VA-3	26.0	29.1
VA-4	26.1	26.1
VA-5	22.4	19.5
VA-6	23.9	23.2
WV-1	25.4	22.5
WV-2	26.0	26.8
WV-3	25.5	25.0
WV-4	26.2	27.9
WV-5	23.9	23.8
WV-6	26.1	26.5
WV-7	26.1	23.9
KY	24.4	25.8
OH	25.2	24.9
WV	25.2	26.2
VA	25.6	25.2
All	25.1	25.5
95% Confidence Interval	24.44 - 25.76	24.37 - 26.69

[†] State means exhibit no statistically significant differences ($\alpha = .05$), as determined via Tukey's HSD test.

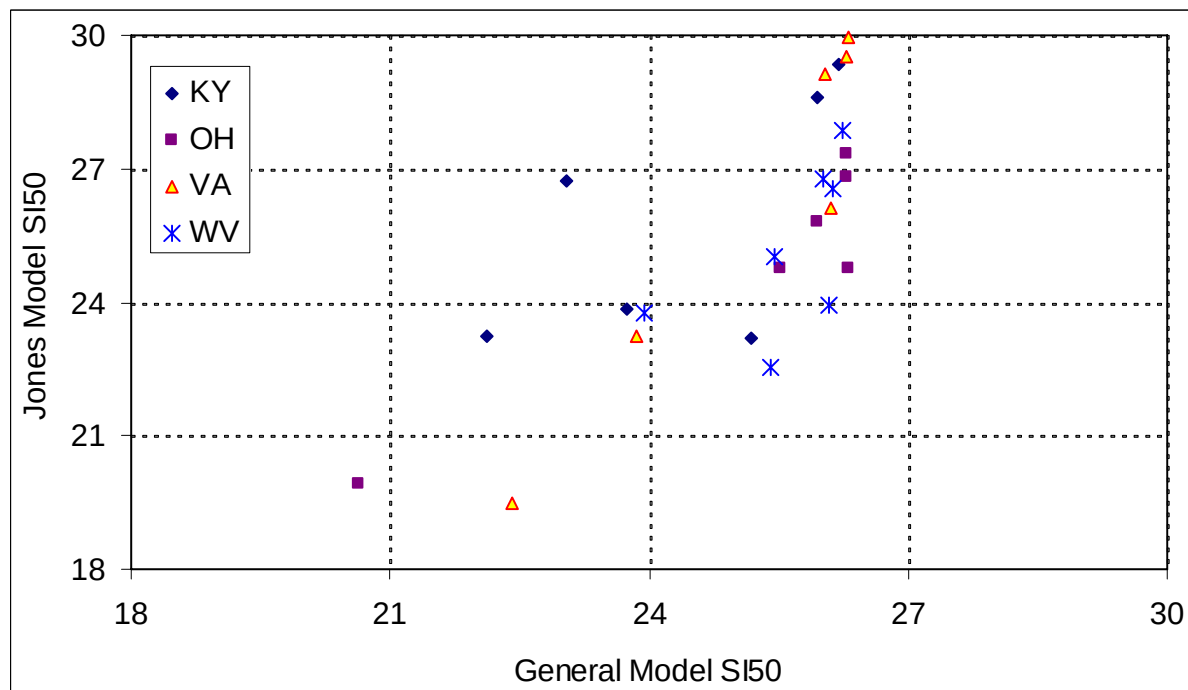
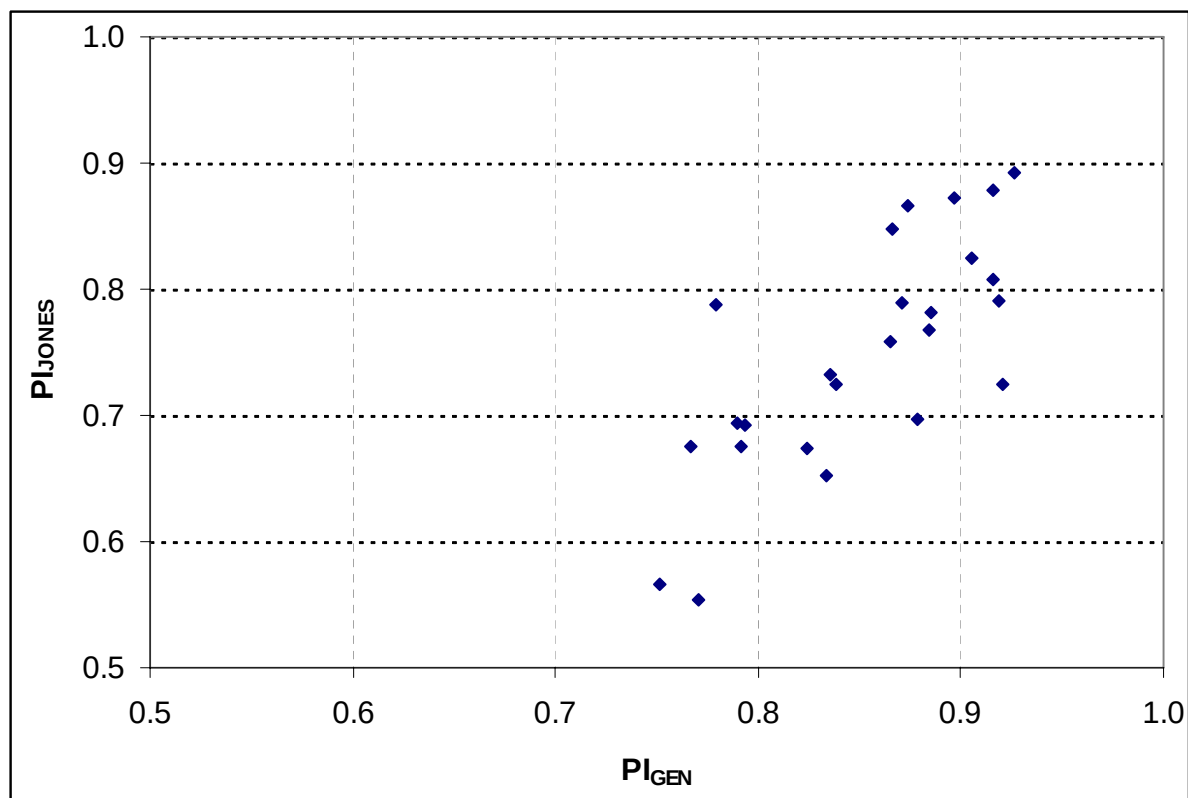


Figure 5-4. Relationship of productivity indices (above) and 50-year site indices (SI50s expressed in meters, below) for mine sites estimated by the Jones Model and the General Model.

Estimating The Potential for Sampled Mine Sites to Accumulate C if Reforested

Equations 5, 6, and 7 were applied to estimate the potential for sampled mine sites to accumulate C over a single rotation if reforested. SI50_{WO} estimates were converted to C sequestration potential estimates for each stand type over 60 years by applying additional assumptions:

- Hardwoods planted for the purpose of sequestering C would be harvested and regenerated after a 60-year rotation, while pines and mixed stands would be harvested and regenerated as two successive 30-year rotations (as per Amichev 2007).
- At the end of an initial rotation: C is distributed proportionately among soil, litter, and biomass components as determined by Amichev on mined sites as per relative proportions measured on mine sites by Rodrigue (2001) and reported by Amichev (2007), while biomass C is distributed among roots and shoots via root/shoot ratio of .22 for pines and .21 for hardwoods, (as per Niu and Duiker 2006; section 2.2.4), and .215 for mixed stands (Table 5-12).
- 87% of above-ground biomass (shoot) C is utilized as carbon-neutral forms – either maintained in non-atmospheric forms over the long term or used as a fossil-fuel substitute energy source (as per Niu and Duiker 2006, Section 2.2.9). This estimate assumes that “best carbon management” practices would be employed at harvest and in disposal of wood-product wastes; that any solid-wood products placed in landfills decompose very slowly; and that landfill CH₄ emissions are not a major issue, as would be likely to occur in response to landfill management changes if carbon constraints were to be imposed.
- Soil C accumulates linearly over the first 60 years, as per assumptions articulated by Heath *et al.* (2002). When this assumption is applied, the mined-land forest soil C pool remain below natural-forest reference levels on comparable sites for the first 60 years after reforestation (Rodrigue 2001, Amichev 2007).
- Essentially 100% of the root C remaining in-ground after first-rotation harvest is subject to decomposition during the second rotation. This assumption is based on application of a decomposition rate constant derived from Silver and Miya (2001) over 30 years.
- Litter C accumulates only over a single rotation and remains relatively constant afterwards. This assumption is supported by Smith and Heath (2002; Table 3), who found residence times of litter-C pools to be in the range of 3-4 years in southern USA, 8-9 years in northern USA, and that typical natural-forest litter quantities for pine and hardwood stands (Smith and Heath 2002, Table 3) are in the range of those being estimated for the mined-land forests at the end of a single rotation; and by the finding by Amichev (2007) that litter accumulations in pine, hardwood, and mixed stands demonstrated no significant statistical relationship with stand age. The estimated mined-land forest litter quantities for single rotations are in the range of those estimated by Niu and Duiker (2006, Fig. 2) for comparable estimation scenarios on afforested agricultural lands.
- For the purpose of this analysis, soil and biomass C accumulation rates were estimated as if the reforestation was being initiated on recently reclaimed mine site (i.e., with zero native C). These “initial reforestation” quantities are adjusted later in the analysis to account for the C accumulation which would accumulate under grass vegetation in the absence of reforestation (baseline levels).

Table 5-12. Mean estimated carbon over a single rotation (30 years for pines and mixed, 60 years for hardwoods) for three stand types for sampled mine sites with site indices estimated using the general model and the Jones model, and all-site averages of the two sets of SI estimates: Total carbon over a single rotation was estimated through application of equations 5-7 and allocated among root, shoot, soil, and litter pools as described in text, referencing Amichev (2007).

	Pines	Mixed	Hardwoods
<i>Proportionate distribution among major pools</i>			
Root/shoot ratio	0.22	0.215	0.21
Roots	14.1%	13.6%	14.2%
Shoots	64.1%	63.5%	67.5%
Soil	7.5%	11.1%	13.6%
Litter	14.3%	11.8%	4.7%
----- (Mg C ha ⁻¹) -----			
<i>General model</i>			
Roots	22.4	20.6	22.3
Shoots	101.9	95.9	106.4
Soil	11.9	16.7	21.5
Litter	22.7	17.9	7.4
Total C	158.9	151.1	157.6
<i>Jones model</i>			
Roots	22.7	21.0	22.3
Shoots	103.3	97.6	106.4
Soil	12.0	17.0	21.5
Litter	23.0	18.2	7.4
Total C	161.0	153.8	157.6
<i>Average of Jones and General Models</i>			
Roots	22.6	20.8	22.3
Shoots	102.6	96.7	106.4
Soil	12.0	16.9	21.5
Litter	22.9	18.0	7.4
Total C	160.0	152.4	157.6

Baseline C accumulation on planted sites (i.e., amounts of C that would have accumulated in the absence of the active intervention that is required to achieve successful reforestation) was estimated through application of the soil organic C accumulation model developed by Akala and Lal (2001) (Figure 5-5).

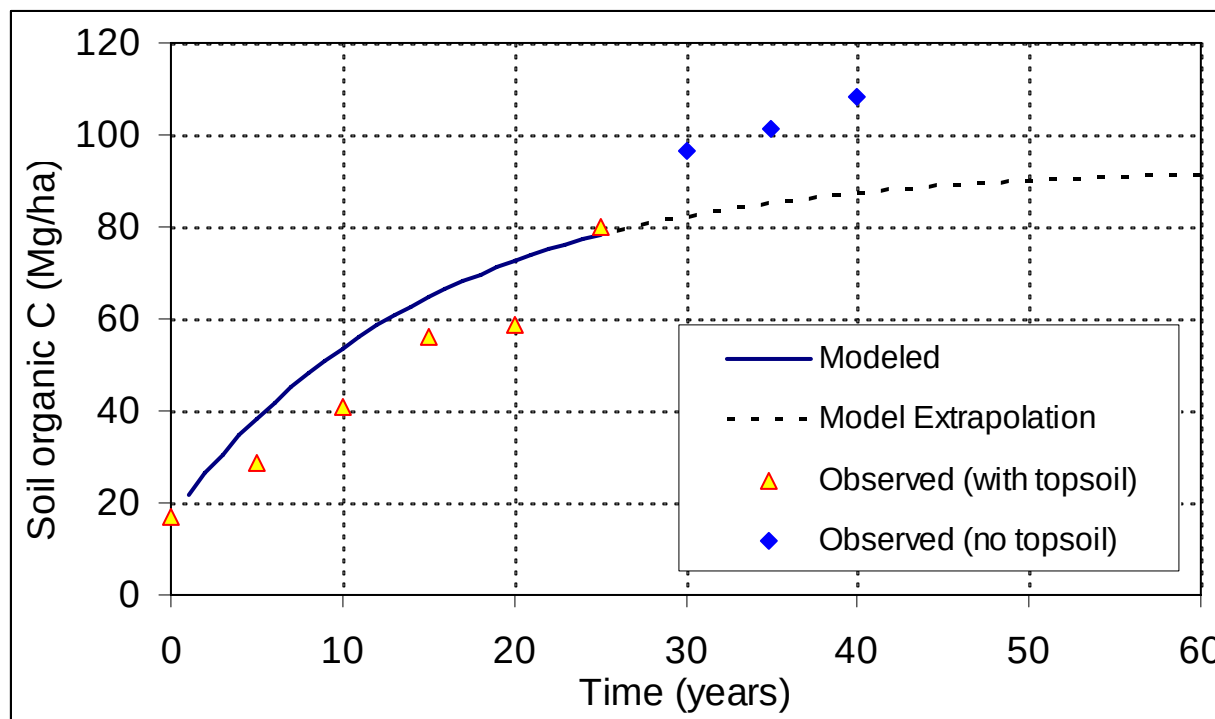


Figure 5-5. Observed soil organic carbon (SOC) accumulation in top 30 cm under grasses in Ohio, on areas reclaimed with and without topsoil, modeled accumulation on the topsoiled areas, and an extrapolation of the model. Observed SOC and the model form are from Akala and Lal (2001). The researchers expressed concern that the SOC in the non-topsoiled plots may have included some residual coal, in addition to pedogenic carbon.

These researchers studied a chronosequence of reclaimed minelands under two land uses, one of which was pasture, in order to determine the potential of reclaimed minesoils to sequester C over time. The rate of SOC sequestration in the pasture treatment ranged from 0.5 to 3.1 Mg ha⁻¹ yr⁻¹ for the 0- to 15-cm depth, from 0.4 to 1.9 Mg ha⁻¹ yr⁻¹ for the 15- to 30-cm depth over the 25-year period reflected by the chronosequence. They modeled sequestration as a function of time over the chronosequence using 2 equations:

$$\text{SOC}_{(0-15 \text{ cm})} = 55 - 45e^{-0.07t} \quad (9)$$

$$\text{SOC}_{(15-30 \text{ cm})} = 38 - 31e^{-0.06t} \quad (10)$$

where SOC = soil organic C (Mg ha⁻¹) within 0-15 cm and 15-30 cm depth-layers, and t = time since reclamation, in years.

We applied the C accumulation model developed by Akala and Lal (2001) to estimate soil-C baseline accumulation on coal-mined sites maintained as grass-dominated herbaceous vegetation (Fig. 5-5). Application of these equations yields an estimate of 93 Mg C ha⁻¹ accumulation over 60 years. However, the coal-mined sites that were the subject of this research

had received an application of topsoil during reclamation. The C contents of topsoils applied on the older sites were not measured at the time of application, but sites being reclaimed contemporaneously with Akala and Lal's (2001) measurements contained 17 Mg C ha^{-1} . Also, the researchers noted that C quantities found in adjacent plots which had not been topsoiled were in the range of $97\text{--}108 \text{ Mg C ha}^{-1}$; however, they also noted a likelihood that these estimates reflected the presence of coal fragments as well as biogenic C accumulation. Our measurements indicated herbaceous biomass C accumulations generally ranged from $1\text{--}2 \text{ Mg ha}^{-1}$, an amount that can be considered as minimal relative to soil C levels. Considering that Akala and Lal (2001) 25-year soil C content is closer to the 30-year (and potentially coal contaminated) measurement than the 20-year measurement (Fig. 5-5), we assumed that coal-mined sites reclaimed with grasses but not reforested would accumulate 75 Mg C ha^{-1} over 60 years, primarily as soil C.

Application of equations 5, 6, and 7 yielded estimates of the total C that would accumulate on each of the sampled mine sites over a single rotation (30 years for pines and mixed stands, 60 years for hardwoods) using the site indices generated by both the General Model and the Jones Model (Table 5-12). These single-rotation total C estimates were allocated among root biomass, above-ground biomass ("shoots"), soil organic, and litter components as described above and averaged for all sites (Table 5-12). Application of the above assumptions regarding amounts of C that can be converted to wood and fuel products or maintained within the forest system through initial rotation(s) over 60 years yielded the "accumulated C" estimate of Table 5-13; application of the assumption that 75 Mg C ha^{-1} would accumulate in the absence of reforestation yields the sequestration estimates (Table 5-13).

Few bases for comparison were found for the estimated C accumulation and sequestration estimates (Table 5-14), but our estimates do appear to be consistent with other references. Our estimates are in the upper portion of the range, but well below the maximum, estimated by the Intergovernmental Panel for Climate Change (2001) for afforestation and reforestation projects globally. Our accumulation- and sequestration-rate estimates exceed those listed by US Department of Energy as reasonable expectations for US afforestation and reforestation projects, which are based on a publication by Birdsey et al. (1996). However, the DOE estimates assume much longer rotations while forest growth rates typically decline with time beyond the optimal harvest period. Our pine- and mixed-stand accumulation-rate estimates are slightly higher than to the 20- and 40-year estimates, and substantially higher than the 80-year estimates, of Galang et al. (2007) for afforestation of marginal agricultural lands in Virginia. However, Galang et al. (2007) estimated only the biomass C (both roots and shoots) that would accumulate in an unharvested forest stand over each time period, as they made no assumptions regarding the proportion of accumulated C that would be "carried over" from one rotation to the next. Galang et al. (2007) assumed that reforestation species selection would be based on suitability for the soil types found on marginal agricultural lands; based on this assumption, about 75% of the afforested acres would be planted with conifers (pines) and 25% with hardwoods. Our C accumulation rate estimates for pines and mixed stands are comparable to the 50-year estimates of Niu and Duiker (2006) in their study of potential afforestation of marginal agricultural lands in 7 midwestern states (including OH, PA, and WV), but our rate estimates for hardwoods are well below those developed by these researchers. Niu and Duiker (2006) considered C sequestration to consist of incremental C accumulation in soil, litter, and biomass on marginal agricultural lands should such lands be afforested. Of the 4 sets of potential comparisons in

Table 5-14, the Niu and Duiker (2006) rates are the most appropriate comparison to our sequestration-rate estimates.

Table 5-13. Mean estimated C accumulation and sequestration after reforestation over 60 years for three stand types on average sites, assuming productivity is adequately represented by averaging General Model and Jones Model estimates for all sampled mine sites. All quantities other than rates as are expressed as Mg C ha⁻¹; rates are expressed as Mg C ha⁻¹ yr⁻¹. Additional detail on assumptions is provided in text.

	Pines	Mixed	Hard- woods	Assumption
C Accumulation				
<i><u>First Rotation</u></i>				
				30 years for pines and mixed, 60 years for hardwoods
Roots	0.0	0.0	22.3	100% decomposition over 30 years after harvest.
Shoots	89.3	84.2	106.4	87% of pines & mixed are transferred to sequestrable forms
Soil	12.0	16.9	21.5	
Litter	22.9	18.0	7.4	Accumulates over 1 rotation only
<i><u>Second Rotation</u></i>				
Roots	22.6	20.8		Living roots at end of rotation
Shoots	102.6	96.7		Standing stock at end of rotation
Soil	12.0	16.9		Linear accumulation continues
Litter	0.0	0.0		No additional accumulation after 1st rotation
<i><u>Over 60 Years</u></i>				
Roots	22.6	20.8	22.3	
Shoots	191.8	180.9	106.4	
Soil	23.9	33.7	21.5	
Litter	22.9	18.0	7.4	
Total C	261.2	253.5	157.6	
Total C Rate	4.4	4.2	2.6	
Baseline C	75.0	75.0	75.0	Estimated soil C that would accumulate over 60 years under grass without reforestation
C Sequestration				Sequestration = C accumulation - "Baseline C"
Total	186.2	178.5	82.6	
Rate	3.1	3.0	1.4	

Table 5-14. Carbon accumulation and sequestration quantities and rates estimated by others.

Estimate	Years	Quantity (Mg ha ⁻¹)	Rate (Mg ha ⁻¹ yr ⁻¹)	Source
<i>Representative C sequestration rates and saturation periods, USA.</i>				Birdsey, 1996
Afforestation	90 - 120		0.6 - 2.6	
Reforestation	90 - 120		0.3 - 2.1	
<i>Total C accumulation in reforestation and afforestation projects, global.</i>				
High		328		IPCC, 2001;
Low		26		Table 5.
<i>Estimated average C accumulation in biomass on afforested agricultural lands in Virginia, USA, single rotations of 3 lengths.</i>				
	20	70.8	3.5	Galang et al.,
	40	109.2	2.7	2007
	80	140	1.8	
<i>Total estimated C accumulation on afforested marginal agricultural lands in 7 midwestern states, USA.</i>				
Conifers - Short Rotation	20	84	4.2	Niu and Duiker
Deciduous - Short Rotation	20	78	3.9	2006; Table 5.
Conifers - Permanent	50	165	3.3	
Deciduous - Permanent	50	160	3.2	
Conifers - Short Rotation	50	160	3.2	
Deciduous - Short Rotation	50	155	3.1	

Estimating Cumulative Ecosystem C Potentials of the Mined Land Resource Base

Several assumptions were made in order to estimate the total C-sequestration potentials of the post-SMCRA mined land resource base:

Assumption 1: The forest productivities of mine sites sampled and characterized are representative of those which occur across eastern US.

We make this assumption out of necessity, given the fact that no alternative assumption is available.

The assumption is supported by the fact that the mean SI50 values derived through application of two models were in close agreement (Table 5-11). The mean SI50_{WO} derived through application of the Jones Model was 25.53 m, and while the corresponding mean SI50_{WO} derived through application of the General Model was 25.10. The 95% confidence intervals for the mean-value estimates are relatively narrow: ± 0.66 m (2.6%) for the General Model, and ± 1.16 m (4.5%) for the Jones model. No significant differences in mine-site mean SI50_{WO} values

were found to occur between states for either model, indicating that it is appropriate to characterize the productivity of eastern US mine sites using data collected from the sampled mine sites, without stratifying by state.

The four states where sampling sites were located accounted for 78% of the combined Phase III and recent (2001-2005) Phase I bond-released lands in eastern US (Table 5-15). The fact that few statistically-significant differences in soil properties, and no statistically significant differences in site index, were found to occur by state supports the assumption that these sites can be considered as at least somewhat representative of eastern US mined lands, and they provide no reason to stratify the analysis by state.

Table 5-15. Eastern U.S. coal-mined land areas reclaimed under SMCRA, 1978-2005.^a

State	Phase III Released	Phase I Released (2001-2005)	Total
	----- (hectares ^b) -----		
Eastern KY ^c	247,176	26,094	273,269
MD	5,372	118	5,490
OH	74,167	9,495	83,661
PA	93,670	13,360	107,030
TN	14,962	2,946	17,907
VA	37,076	1,126	38,202
WV	93,686	11,674	105,360
Total	566,107	64,813	630,920

^a including the interim SMCRA program. Source: US OSMRE “20th Anniversary of the Surface Mining Law” (<http://www.osmre.gov/annivrep.htm>) and annual reports to Congress.

^b As reported by states to OSMRE; these figures overestimate total affected areas due to double-counting of areas that were both mined and re-mined under SMCRA.

^c Estimated from total Kentucky areas, as proportionate to the east-west distribution of surface coal tonnage.

Assumption 2: The quantity of Eastern U.S. mined lands that could be available for reforestation and sequestration is on the order of 300,000 ha.

This estimate was derived assuming approximately half of lands mined and reclaimed under SMCRA could be made available for C sequestration. The 300,000 ha figure is slightly less than half of the total Phase 3 and recent Phase I (2001-2005) bond released acreage tallied by US Office of Surface Mining (Table 5-12). The Table 5-12 acreages include some double-counting due to remining of acreages that were mined previously under SMCRA. We were unable to locate any data to support an estimate of doubly-counted acreages, and we were unable to persuade any of the state-agency officials that we discussed this issue with to offer an informed estimate even for their own state. Therefore, we are estimating total post-SMCRA mined acreage at 600,000 ha, less than the 630,000 ha total of Table 5-12, as a means of adjusting for double-counting due to remining. We also note that bond-release acreage totals include deep-mine face-ups, refuse piles, roads and ponds, and similar areas that were not “surface mined” in any conventional sense

We are estimating that only 50% of available acreage could be made available under the right incentive structure considering several factors. First of all, some of the post-SMCRA mined lands are already in beneficial uses, such livestock grazing, and various forms of development. Two of the 25 sites we visited were planted with forest trees at the time of mining. We are aware of no data to allow an estimate of such a quantity, but our experience indicates it to be very small, far less than 50%. Second, some landowners may not wish to make their lands available for C sequestration even if additional incentives were offered. If mineral owners believe that future marketplace changes will make it profitable to mine such properties again, so as to extract coals that that could not be exploited economically when past mining was occurring, most would consider potential returns from future mining as likely to far outweigh potential returns from reforestation and C sequestration. However, we would not expect a majority of landowners to fall into this category.

A third factor considered concerns the status of the vegetative communities on mine sites at present. Assuming soil properties are equally favorable, reforestation will be most feasible where mine sites are not occupied by invasive woody vegetation (such as the autumn olives that are proliferating in VA and WV mining areas) or by low-value species with poor timber-production and C-sequestration potentials that may have been planted after mining (such as black locust, which has been used commonly for reclamation in VA, WV, and KY). If such species are present and have advanced to the point where they would hinder reforestation success (by interfering with equipment operation, or by shading and out-competing planted seedlings), they must be eliminated if the site is to be successfully reforested; this can only occur at some cost. For example, we are aware of a Virginia site that is being considered for reforestation despite being occupied by a mix of planted and invasive woody species that must be removed if the reforestation is to be successful; this community is thick and well advanced, with some individuals in excess of 15 feet tall. The estimated cost for removing this community, so as to enable successful reforestation to occur, is on the order of \$2400 per hectare, well in excess of what can be supported economically considering potential reforestation and C sequestration concerns. About $\frac{2}{3}$ of the cover provided by dominant and co-dominant vegetation types at sampling locations was provided by planted herbaceous species and native herbaceous species, vines, and shrubs, vegetative communities that we interpret as consistent with conditions necessary for successful reforestation.

Assumption 3: That post-reforestation C sequestration on planted sites can be estimated using the method described above.

A weak link in this assumption concerns the use of mined lands that are maintained in herbaceous vegetation as the reference for determining baseline C accumulation, given that forest succession does occur on eastern US mined land sites, albeit at a much slower rate than would occur through active reforestation of these sites. In our experience, it is apparent that reclamation with aggressive herbaceous species such as tall fescue and sericea lespedeza retards natural succession on mine sites; however, we cannot state with certainty that such an effect would be maintained under unmanaged conditions over 60 years in the absence of active reforestation. Since we have no way to estimate C accumulation rates under a “natural succession” scenario, we have used grassland soil C accumulation rates as to estimate baseline C quantities.

Assumption 4: An immediate and rapid effort can be mounted to provide incentives for post-SMCRA mined land reforestation.

This may or may not be a realistic assumption, but it allows us to conduct our analysis using year 2010 as a temporal baseline and consider the average midpoint of the 60-year period modeled by equation 6 to be 2040.

C Sequestration on Post-SMCRA Mined Lands

In order to estimate the C sequestration potentials of post-SMCRA mined lands, we will use the “mixed stand” sequestration-rate estimates of 178 Mg C ha⁻¹ over 60 years (Table 5-13). Use of mixed stands, as opposed to a reforestation strategy that would maximize sequestration by relying solely in pines, would result in only a slight reduction of total C sequestered. However, given that the region’s native forest vegetation is predominantly hardwoods, reliance on mixed stands could be expected to provide a higher level of ancillary benefits than reliance on monoculture pine plantings.

Assuming that 300,000 ha could be reforested with an appropriate incentive structure allows extension of the above to estimate that 53.5×10^6 Mg C could be sequestered on eastern US mined lands over 60 years, an average of 3.0 Mg C ha⁻¹yr⁻¹.

Determine Significance of Potential C Sequestration on Mined Lands

The potential significance of these lands’ C sequestration potentials was estimated assuming lands would be reforested over a multiple-year period centered on the year 2010 (Table 5-16). Since these lands’ C sequestration potentials were estimated for a 60 year periods, the mid-point of that 60-year period (2040) was selected as a reference for use in determining the significance of estimated annual average mined-land sequestration rates. Total US energy-related C (as CO₂) emissions are projected by US Energy Information Administration (EIA 2006) for the year 2030 (“Reference Case” forecast, Table A2); a 2040 “business as usual” US emissions estimate was developed by assuming the 1.2% annual increase projected for the 2004-2030 period by US EIA (2006) would continue forward until 2040.

Table 5-16. Summary of key assumed quantities, C emission reference levels, and findings: Carbon sequestration potentials on eastern US coal-mined lands

300,000:	ha of lands mined for coal and reclaimed under SMCRA available for reforestation (assumption – see text)
25.3:	Average 50-year site index of mined sites for white oak, expressed in feet.
253.5:	Mg C ha ⁻¹ = average potential for C accumulation on reforested mine sites, over 60 years (Table 12, Mixed Stands).
75:	Mg C ha ⁻¹ = average C accumulation on reclaimed mines over 60 years, in the absence of reforestation (equations 7 and 8).
178.5:	Mg C ha ⁻¹ = average net sequestration potential due to reforestation, over 60 years (Table 13).
2.98:	Mg C ha ⁻¹ yr ⁻¹ = average annual net sequestration potential due to reforestation, over 60 years..
53,550,000:	Mg C = net sequestration potential on mined lands over 60 years, as per above assumptions.
892,500:	Mg C yr ⁻¹ = average annual net sequestration potential on mined lands over 60 years, as per above assumptions.
1.36:	x 10 ⁹ Mg C emissions from energy production, USA 1990
1.61:	x 10 ⁹ Mg C emissions from energy production, USA 2004
2.21:	x 10 ⁹ Mg C - Projected “business as usual” emissions from energy production, USA 2030 (US EIA 2006, Reference Case Projection)
2.49:	x 10 ⁹ Mg C - Projected “business as usual” emissions from energy production, USA 2040 (assume 1.2% annual increase, 2030 - 2040)
0.04%:	Average annual mined lands C sequestration potential as a proportion of 2040 projected US C emissions from energy production.
1.27:	x 10 ⁹ Mg C = US Kyoto-based C-emissions target (93% of 1990 emissions)
1.23:	x 10 ⁹ Mg C = Emissions reductions from projected "business as usual" 2040 emission levels, that would be necessary to achieve a Kyoto-based target.
0.07%:	Average annual mined lands C sequestration potential, as a proportion of 2040 emission reductions necessary to achieve a Kyoto-based target.
19.0%:	Regional (KY, MD, OH, PA, TN, VA, and WV) C emissions, as a proportion of US total in 2001.
4.5%:	Regional (KY, MD, OH, PA, TN, VA, and WV) C emissions, as a proportion of world total in 2001.

Consideration of the above assumptions leads to an estimated ~890,000 Mg C yr⁻¹ that could be sequestered on eastern US mined lands, on average, over a 60-year period. This amount is equivalent to about 0.04% of the USA’s projected “business as usual” energy-related C emissions over the period. Should the USA adopt a C-emissions offset/reduction target of 7% of 1990 emission levels (which is consistent with targets adopted by parties participating in the

Kyoto Protocol), eastern US post-SMCRA mined land reforestation would be capable of sequestering, on average, amounts of C equivalent to 0.07% of the offsets and reductions that would be required to meet that Kyoto-based target in 2040, our reference date. Should C emissions reductions targets be adopted and distributed regionally, the regional significance of mined-lands' sequestration potentials can be considered in light of the fact that C emissions by the seven eastern coal-mining states considered by this study (KY, MD, OH, PA, TN, VA, and WV) were responsible for 19% of US C emissions in 2001. By assuming that the recent ratio of regional to national emissions would remain constant, we can calculate that annual mined-land sequestration potentials would be capable of offsetting about 0.2% of regional emissions on 2040. By assuming that national targets might be pro-rated regionally in as ratio to recent emissions, we can calculate annual mined-land sequestration potentials as equivalent to about 0.4% of a pro-rated 2040 Kyoto-based target.

However, an alternate approach to implementation would be as one component of a strategy to achieve C-emission offsets and reductions by applying multiple mechanisms. For example, Pacala and Socolow (2004) have proposed a climate stabilization strategy that uses existing technologies to stabilize global C emissions at approximately current levels, 7 Gt C yr⁻¹. Under a "business as usual" scenario (i.e., the absence focus on C emissions), annual global emissions are expected to increase from 7 to 14 Gt C yr⁻¹ over the next 50 years. Pacala and Socolow (2004) propose that climate stabilization be achieved by dividing the offsets and reductions necessary to stabilize effective emissions at current levels into seven equal segments (called "stabilization wedges" – see Figure 5-6), each to be achieved by using available technologies and "ramping up" emissions reductions linearly from zero at outset to 1 Gt C/yr (a total of 25 Gt) over a 50-yr period.¹⁰ U.S. energy-related C emissions constituted 23.8% and regional C emissions constituted 4.5%, of global emissions in 2001. If the Pacala and Socolow (2004) strategy were adopted with U.S. agricultural and forest management activities assigned pro-rata responsibility for achieving one stabilization wedge (5.9 Gt, calculated as 23.8% of 25 Gt, over 50 years), eastern U.S. mined lands' C sequestration (~45 x 10⁶ Mg over 50 years, calculated as 5/6 of the 53.5 x 10⁶ Mg 60-year estimate) would be capable of achieving about ~1% of that target. Should responsibility for meeting that target be distributed regionally in proportion to recent emissions, reforestation of mined lands could be capable of meeting about 3-4% of the eastern states' (KY, MD, OH, PA, TN, VA, and WV) pro-rata share of that stabilization wedge.

Thus, under almost any scenario, the amount of C that could be sequestered on eastern U.S. coal-mined lands would be inadequate as standalone strategy for offsetting energy-related C emissions.

¹⁰ 1 gigatonne (Gt) = 10⁹ megagrams (Mg)

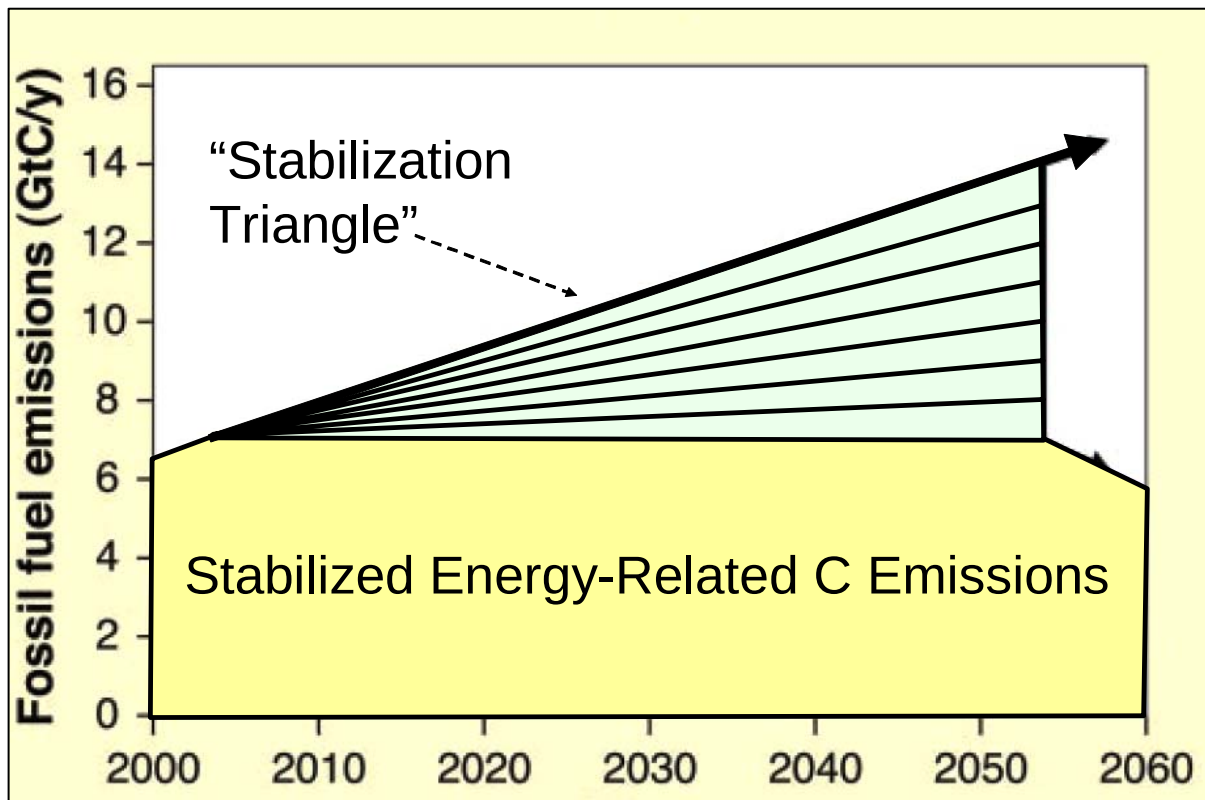


Figure 5-6. A conceptual model of the “stabilization triangle” concept advanced by Pacala and Socolow (2004). The “stabilization triangle” is comprised of seven “stabilization wedges,” each of which would reduce or offset C emissions at rates increasing linearly from zero at outset to 1Gt C / yr over a 50-year period. Figure adapted from Pacala and Socolow (2004).

Comparison to Other Estimates

Three other studies have attempted to estimate the C sequestration potentials of the US mined land resource base (Table 5-17). Those studies applied very different assumptions reached conclusions that differ from those of the current study. However, because of differences in assumptions and methods, these researchers' findings are not directly comparable to those of the current study.

Table 5-17. Carbon accumulation / sequestration potentials of the U.S. mined land resource base: Comparison of this study's results to other estimates.

Study	Total (10^6 Mg C yr ⁻¹)	Time (yrs)	Area (ha)	Average Rate (Mg C ha ⁻¹ yr ⁻¹)	Notes
Current study	0.90	60	300,000	3.0	Approx. half of eastern US post-SMCRA coal surface mined lands are available for reforestation.
Amichev 2007	2.50	60	650,000	3.8	All eastern US post-SMCRA coal mined lands are available for reforestation. Estimate includes only C accumulation on mined lands, is not adjusted to account for "baseline" (without active reforestation) accumulation)
Shrestha and Lal 2006		n/a	3,200,000		All US mined lands are available for "restoration of ecosystem function," and all currently permitted acreages will eventually become available.
Upper bound	3.20			1.0	Soil Organic C only; not adjusted to account for baseline.
Lower bound	1.60			0.5	Soil Organic C only; not adjusted to account for baseline.
Ussiri and Lal 2005 (average)	1.28	25	1,250,000	1.0	All US coal-mined lands (both pre- and post-SMCRA) are available for use in carbon sequestration. Soil Organic C only; not adjusted to account for baseline.

Amichev (2007, p. 283) estimated the amount of C that could be accumulated if all eastern US post-SMCRA coal-mined lands (650,000 ha) were reforested with mixed hardwood and pine species. He estimated these lands' total C accumulation potential as 2.5×10^6 Mg C ha⁻¹ yr⁻¹ on average over the full 60-year rotation, approximately 2.5 times the current study's estimate.

Shrestha and Lal (2006) estimated the potential capacity of US mined lands to sequester

atmospheric C as soil C at 1.6 to 3.2 Tg yr⁻¹ (Shresthra and Lal 2006).¹¹ However, these researchers estimated the area available for use C sequestration activities as 3.2 x 10⁶ ha, based on the total land area permitted for coal mining under SMCRA in 2003 assuming that all such land would be mined and reclaimed. These researchers included only soil C in their estimates.

Ussiri and Lal (2005) estimated the capacity of U.S. mined lands to sequester atmospheric C as soil C over 25 years as ranging from 6.25 to 57.8 Tg C over 25 years, averaging 32 Tg C or 1.28 Tg C yr⁻¹. These conclusions were based on the assumption that the total area of land mined for coal through 2002 (including both pre- and post-SMCRA mining, a total of 1.25 x 10⁶ ha) could be available for C sequestration. These researchers included only soil C in their estimates.

In contrast to these other studies, the current study assumed:

- That not all lands mined for coal are available for reforestation. We estimate the total area of land that has been mined under SMCRA and is available for reforestation / C sequestration at about 300,000 ha, assuming that roughly 50% of the total area could be made available. We consider the 630,000 ha of bond-released lands to be an overestimate of the total surface-mined area, since this figure includes some double-counting (areas mined under SMCRA in its early years, and then remined at a later date with the larger equipment that has become available), as well as deep-mine face-ups, refuse piles, roads and ponds, and similar areas that were not “surface mined” in any conventional sense. Far less than 100% of surface-mined lands are available for reforestation, given that some of these were reclaimed on forested post-mining land uses prior to bond release, others are currently in alternative uses (pasture, commercial, industrial, etc.), and that some landowners of post-SMCRA mined lands that are not in alternative use will not be willing to devote lands to the permanent forest cover so as to aid in offsetting climate change – especially if potentially remineable coals remain in place on the property. We have no factual basis beyond our own experience for estimating the available-for-reforestation proportion of surface mined lands at roughly 50%; that value may or may not be close to correct.
- That other studies did not adjust for “baseline” C, i.e., that amount of C that would be stored by vegetation and soils in the absence of active reforestation. Only when C accumulates as a response to management actions in quantities over and above that which would accumulate in the absence of such management can that C be considered as “sequestered” a condition commonly termed as “additionality.”

Our own adjustment for additionality used C accumulation by lands in grasses as the baseline quantity, despite the fact that many of the mine sites sampled showed evidence of natural succession through invasion of shrubby and low-value woody vegetation. We would expect that C accumulation rates on such sites to be far slower than on sites that have been actively reforested, but we have no data to estimate what those rates might be. Because Akala and Lal (2001) contained measurements of long-term carbon accumulation on grassland sites, we used these data as the baseline as needed to estimate C accumulation rates on reclaimed coal-mined sites.

¹¹ 1 teragram (Tg) = 10⁶ megagrams (Mg)

Summary and Conclusions

Eastern U.S. lands that have been mined for coal and reclaimed under the Surface Mining Control and Reclamation Act constitute a significant land resource. U.S. Office of Surface Mining records indicate that nearly 600,000 hectares of such land have been reclaimed under SMCRA in a manner that satisfies all legal obligations and thus have been released from regulatory oversight, while additional acreages, still under regulatory oversight, have been mined and reclaimed. Large proportions of eastern US coal-mined lands have not been reforested and are currently not under active management for any economically valued purpose. We investigated the potential for reforestation of these lands for the purpose of sequestering atmospheric C. Should such reforestation occur, it would also produce ancillary benefits by providing environmental services, such as enhanced watershed protection, and producing timber and renewable-fuel products.

Soil properties on these mine sites appear to be generally well-suited for reforestation. The mean 50-year site index for white oak on 25 mine sites identified through a random selection procedure was estimated at about 25 meters using two different methods, a level that would be considered as better-than-average on natural soils occurring in most eastern US areas mined for coal. Our study located sampling points using a random selection procedure; thus, sampling points often occurred on sloped areas, especially within the central Appalachian (VA, southern WV, eastern KY) portion of our study areas; soil compaction was not found to be a major limiting factor on most sites, in part because the relatively flat areas that often become compacted by heavy equipment constituted only a minor proportion of our study sites, but soil compaction did appear as major limiting factor to reforestation potential on some of the lowest SI_{50WO} sites. Low amounts of plant-available phosphorous, a soil property that would be easily ameliorated during a reforestation procedure through fertilization, was the property that our study found to be most limiting to reforestation potentials. These findings indicate that many sites could be reforested successfully without engaging in costly physical amelioration procedures, such as ripping.

One major site factor that occurs commonly was not included in the soil productivity models, and that can be expected to hinder reforestation of sites where it occurs is the proliferation of low-value woody species with little potential to produce marketable timber – especially black locust (which has been planted by many mining operations under SMCRA) and autumn olive (a non-native invasive species that was formerly planted by mining operations and is proliferating as its seed are spread by birds and by other means). When mine sites become occupied by such species in advanced growth stages, the resultant woody biomass becomes a barrier to successful reforestation. Their presence acts as a physical obstacle to equipment operations, and, unless removed prior to planting, their rapid growth can be expected to retard planted seedlings growth and survival. When these species reach advanced stages, the cost of removal can render reforestation investments unprofitable. As time passes, it is likely that many mine sites which are not currently occupied by such species in advanced growth stages will become so occupied, and their potential for reforestation will decline as a direct result.

Although there are mine sites throughout the eastern U.S. that possess physical properties that would be ideal for reforestation, their collective potential to sequester atmospheric C appears as minimal relative to current US C emissions. If on the order of 50% of the lands mined for coal and reclaimed under SMCRA in the eastern US were to be reforested, the estimated sequestration potentials of these lands over a 60-year period would be in the order of 975,000

Mg C yr⁻¹, on average, equivalent to about 0.04% of the USA's projected "business as usual" energy-related C emissions in 2040, the midpoint of that 60-year period for lands that that could be if a rapid program were to be initiated in the short term.

Although potential sequestration quantities are not great relative to potential national needs should an energy-related C-emissions offset requirement be developed at some future date, these lands are available and unused for other economically valued purposes, while many possess soil and site properties that are well-suited to reforestation. As time passes, their collective potential for reforestation can be expected to decline due to the continued proliferation and growth of low-value woody species with minimal C sequestration potentials, including non-native invasives that were formerly planted commonly during reclamation under SMCRA on coal surface mines.

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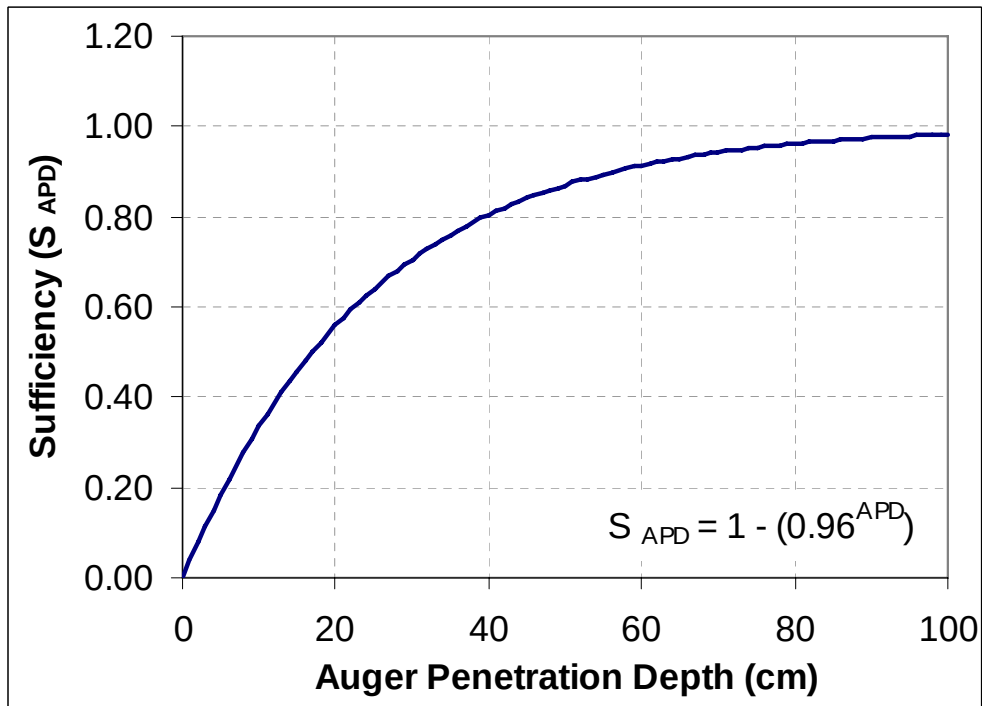
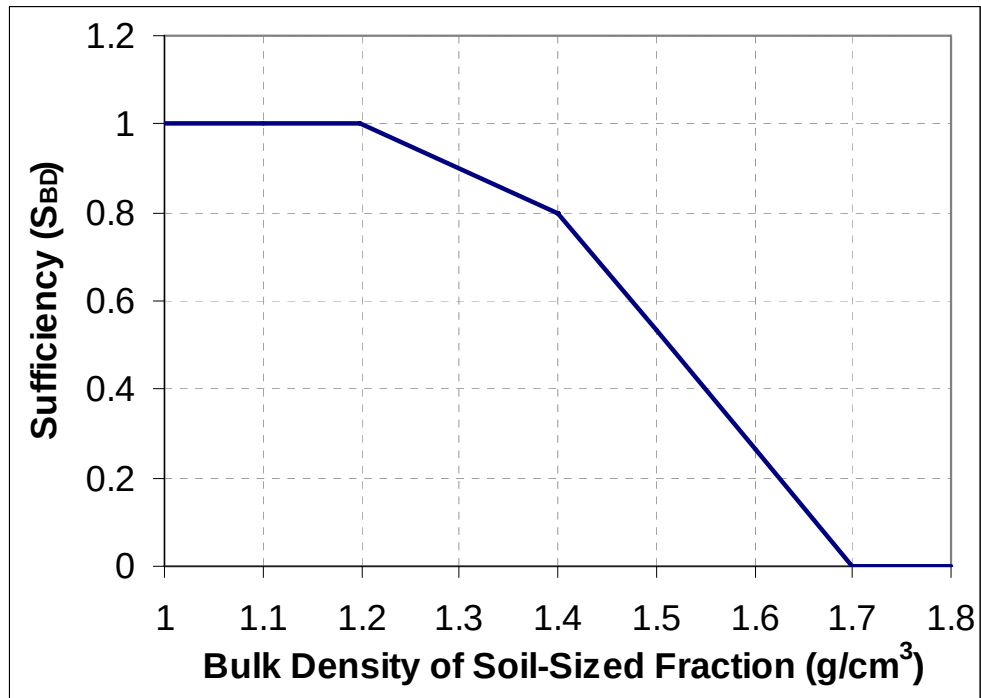
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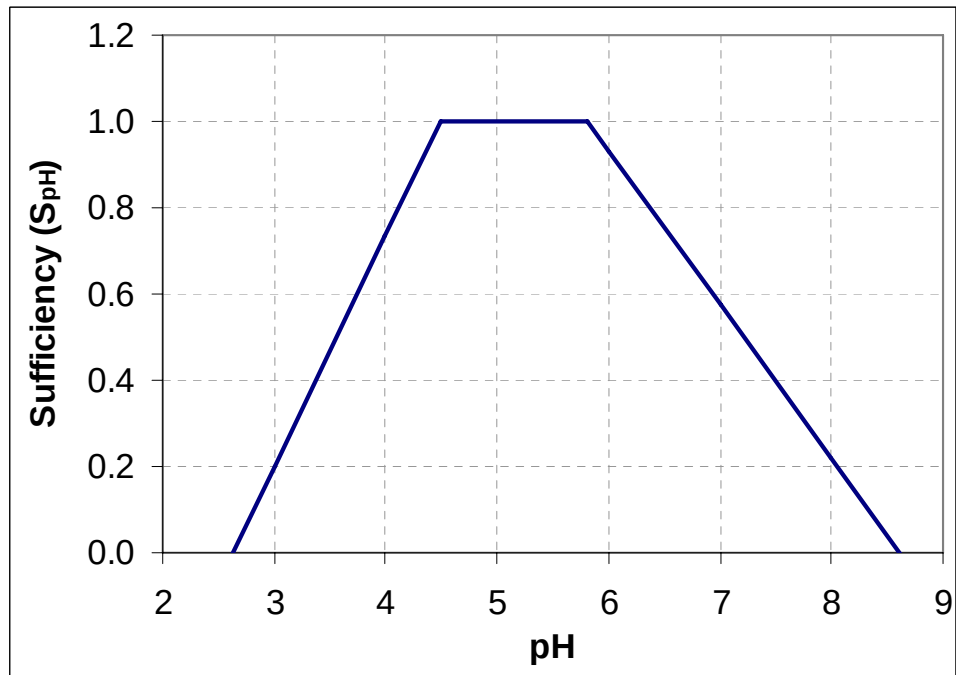
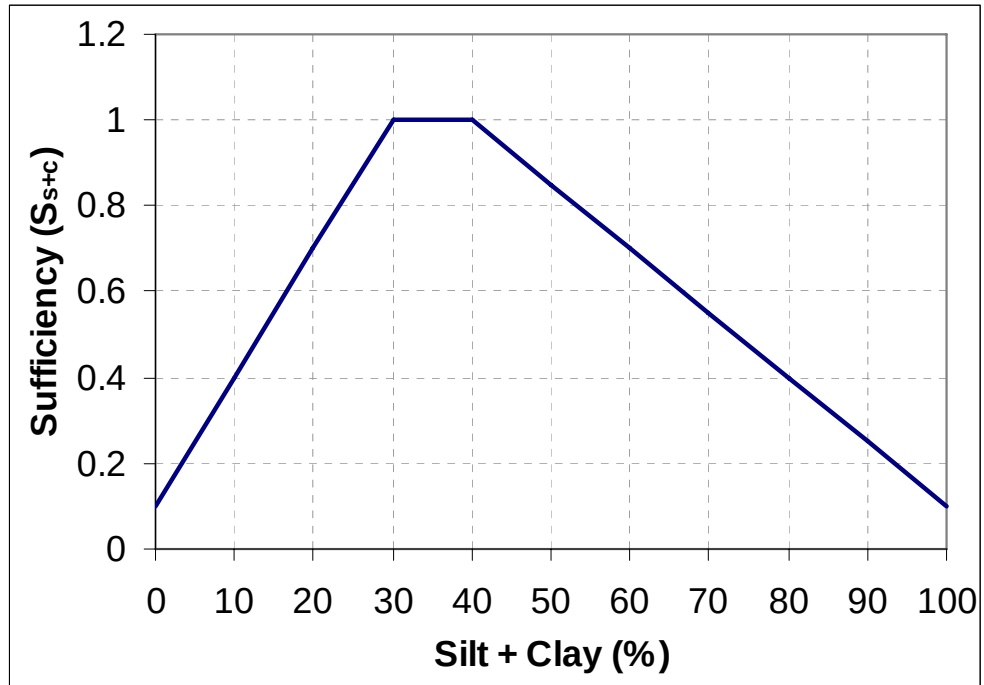
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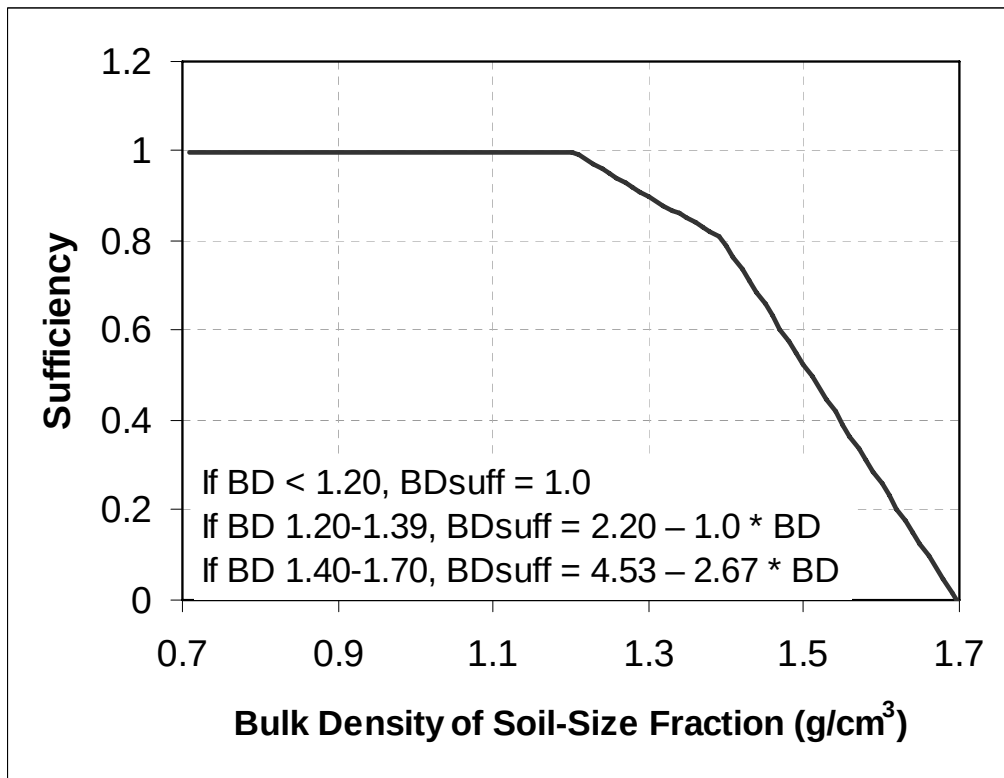
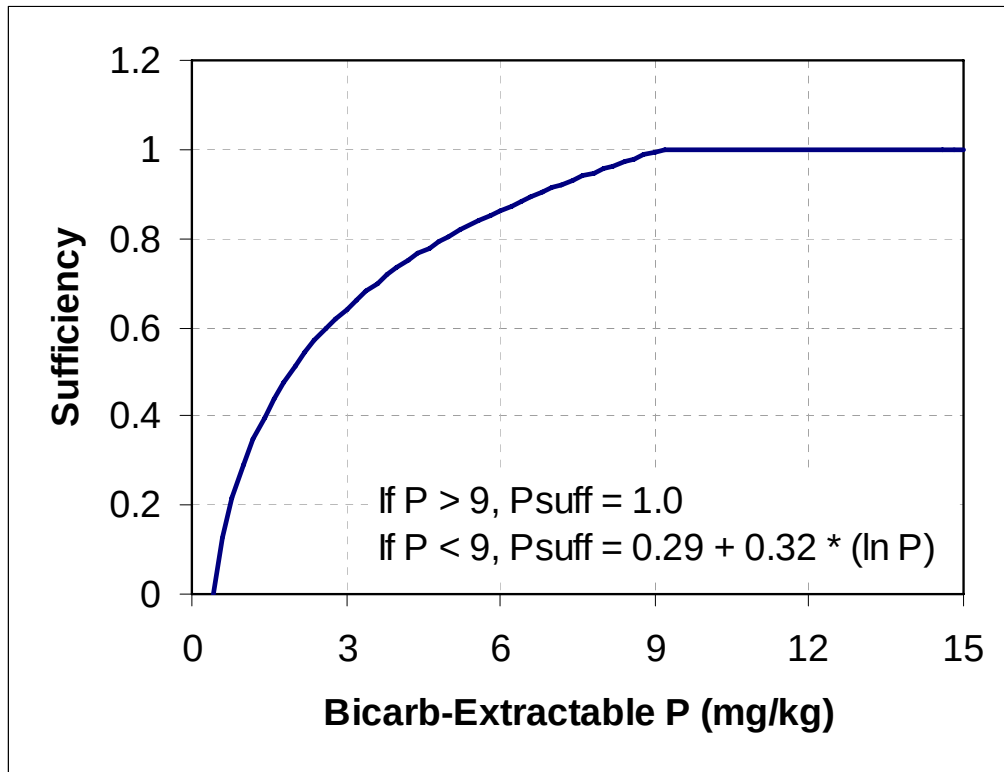
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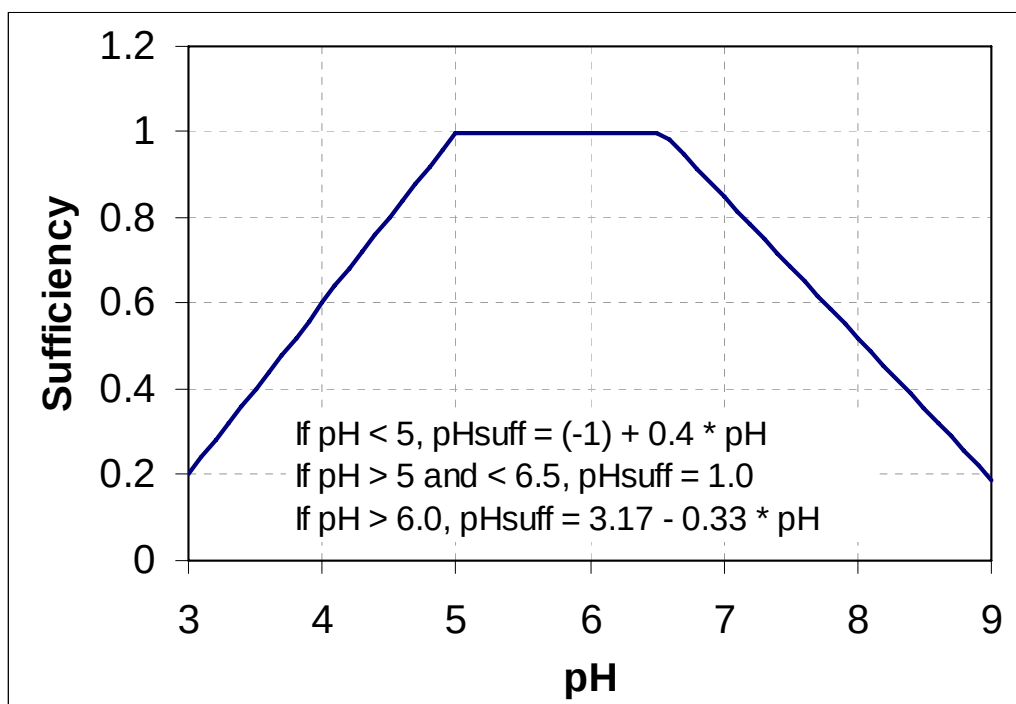
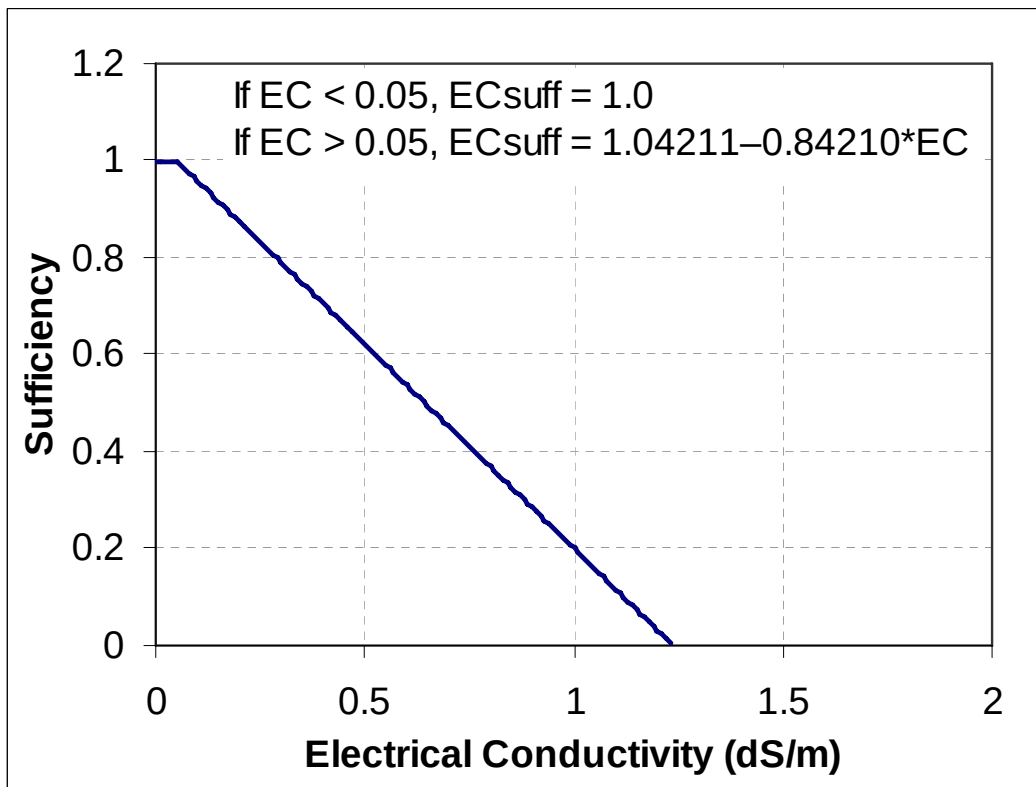
Appendix 5-A. Sufficiency Curves – Jones Model

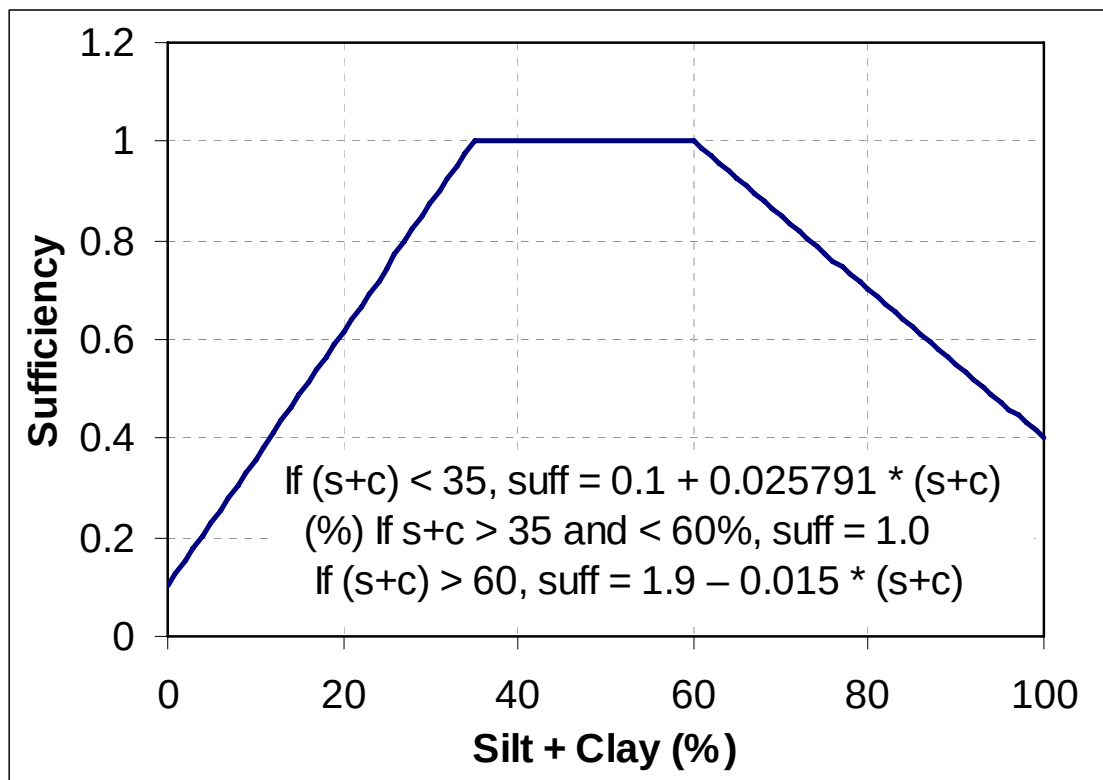
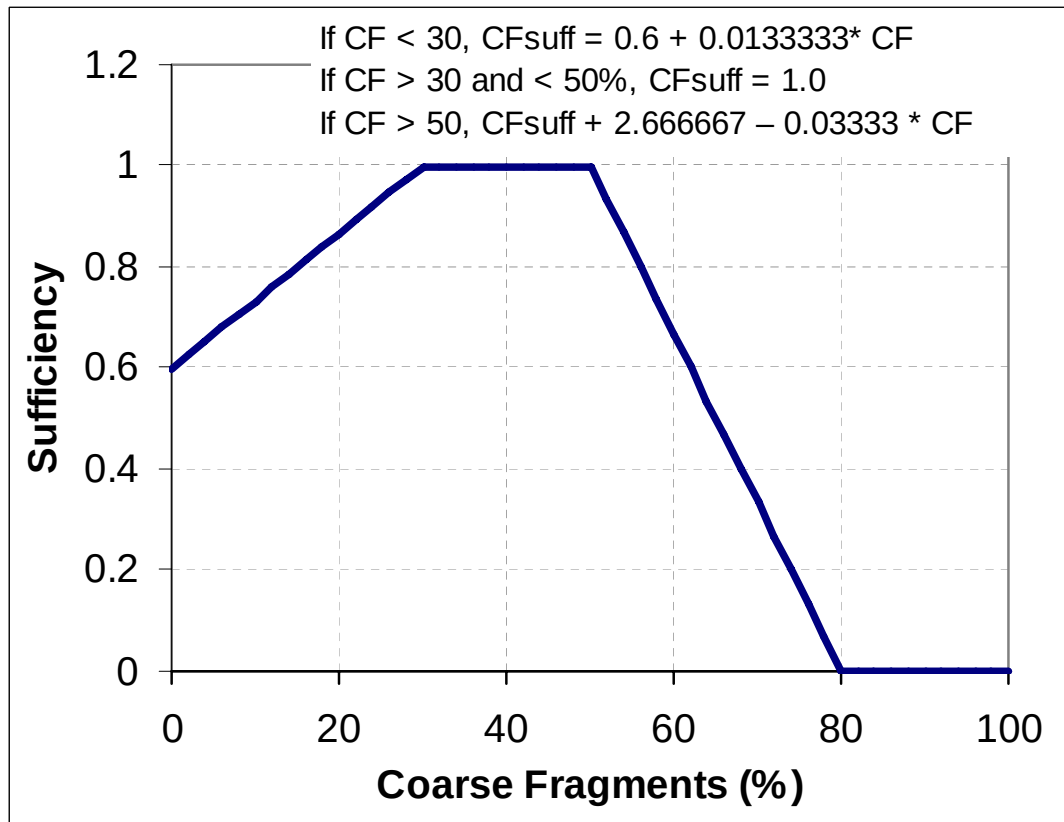


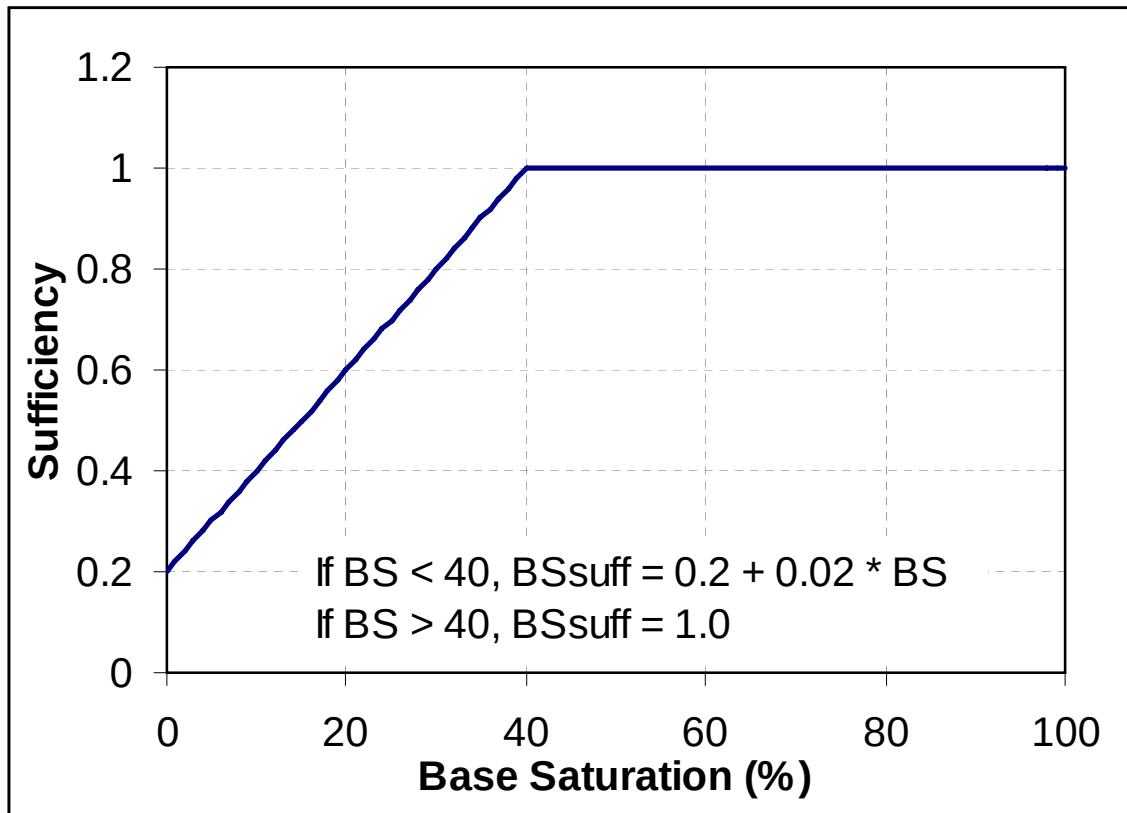


Appendix 5-B. Sufficiency Curves – General Model









PROJECT TIMETABLE

		Planned				Completed						
Year:	2002	2003				2004				2005		
Quarter:	4th	1st	2nd	3rd	4 th	1st	2nd	3rd	4th	1st	2nd	3rd
Task 1												
Subtask 1.1	Baseline Carbon Sequestration Potential											
Subtask 1.2	Mine Soil Productivity											
Subtask 1.3						Carbon Sequestration by Forest Practice						
Subtask 1.4						Accounting Procedures						
Task 2												
Subtask 2.1	Classification Criteria											
Subtask 2.2				GIS Mapping								
Subtask 2.3				Test Remote Sensing								
Subtask 2.4						Experimental Plots						
Subtask 2.5						Soil Analyses						
Subtask 2.6											Validate classification criteria	
Task 3												
Subtask 3.1	Locate sites											
Subtask 3.2			Establish experiment									
Subtask 3.3			Silvicultural recommendations									
Subtask 3.4						Reforestation costs						
Subtask 3.5						Evaluate survival and growth						
Subtask 3.6						Estimate growth potential						
Subtask 3.7						Estimate timber & carbon value						
Task 4												
Subtask 4.1	Economic feasibility											
Subtask 4.2				Evaluation								
Subtask 4.3								Government policies				
Task 5												
Subtask 5.1			Identify SMCRA grassland									
Subtask 5.2				Use characteristics of permits								
Subtask 5.3						Soil properties by permit						
Subtask 5.4								Est. quantity grassland				
Subtask 5.5								Est. C sequ. by site quality class				
Subtask 5.6											Est. C seq. by policy scenario	