Project Final Report

Demonstrate the Potential for Large-Scale Carbon Sequestration by Reforestation of Mined Lands Using Managed Forests

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Executive Summary

Purpose

This project evaluated the feasibility of restoring high-quality forests on mined land, and it measured carbon sequestration benefits that could be achieved from the application of reforestation procedures.

This is important because to achieve national carbon sequestration goals, it is necessary to develop a broad suite of options that have potential to sequester significant amounts of carbon, that 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 removed in the process of mining coal – which is itself used for energy production with consequent CO_2 emissions. Therefore, it is logical to restore coal-mined lands to productive forest systems that will maximize carbon recapture in vegetation, litter, and soil while providing traditional benefits to landowners.

According to Office of Surface Mining estimates, over 1.1 million hectares in the east were under active coal mining permits during 2001; of these lands, approximately 200,000 are currently classified as "disturbed" (OSM, 2002). Although an accurate measure of the total acreage converted from forests to grassland is not available, we believe it is between 150,000 and 170,000 hectares in the eastern U.S. alone.

It is clear that there is great potential for sequestering carbon on reclaimed mined land, but in order to make it happen several things are needed: (1) Productive forests must replace unproductive grassland and scrubland on reclaimed mined land. In order for this to occur, we need techniques for improving mined land quality. (2) Owners of mined land need to be able to project the economic returns to proposed investments intended to enhance the forest productivity of their land holdings. In order for this to occur, mined-land owners must have the capability to assess mined-land productivity, so that site-specific species and management prescriptions can be made. (3) The forestry enterprise must be operationally viable, which means silvicultural techniques are needed for planting and growing native hardwoods and other forest types. (4) Owners of mined land must know the potential of their mined lands to sequester carbon, and the potential for returns from carbon-sequestration markets to augment their timber-production income. In order for this to occur, methods are needed for measuring, monitoring, and projecting carbon sequestration in forests on mined lands across site quality gradients. Therefore, we pursued the following objectives.

Project Objectives

1. Estimate forest productivity and carbon sequestration potential on mined lands.
    a. Determine the primary mine soil properties influencing forest growth.
    b. Determine baseline carbon content of forests and soils on 14 mined and 7 non-mined sites across a site productivity gradient.
    c. Estimate carbon sequestration potential of pines and mixed native hardwoods across a gradient of mine soil quality.

2. Develop husbandry practices for enhancing the potential of mined land to sequester carbon.
    a. Determine the type and extent of soil physical amelioration, weed control, and fertilization needed to renovate compacted sites of varying quality for an acceptable level of forest productivity.
    b. Determine which species, or combination of species, are biologically and economically suitable on mined sites for different forest capability classes.
    c. Develop management guidelines for the reforestation of abandoned grassland.

Results

Mine Soil Quality

The Surface Mining Control and Reclamation Act requires that reclaimed mined sites be capable of the same productive land uses that occurred prior to mining. Our study demonstrated that pre-law mined sites can be as productive as their non-mined counterparts provided mine soils are suitable for trees. Trees planted on reclaimed mined sites in the Midwest prior to the passage of SMCRA grew as productively as before mining. Mining in the East degraded the quality of some sites. Sites with degraded productivity had high coarse fragment contents and shallow depths, preventing trees from exploiting the soil volume to meet their nutrient and water requirements. Improper spoil selection also created growth-limiting chemical conditions such as low base saturation and high salt content.

Soil characteristics that had the greatest effect on tree growth included profile base saturation, profile coarse fragments, profile available water, C horizon total porosity, and profile EC; these properties explained 52 % of the variation in site index. Productive mine sites were commonly well-drained, ungraded mixtures of weathered coarse and fine textured materials. Base saturation was commonly greater than 70%, and coarse fragments averaged 59%. Profile available water averaged 30 cm, and C horizon total porosity averaged 55%. Profile EC averaged 87 $\mu S\ cm^{-1}$.

The soil properties identified by this study represent soil attributes fundamentally important to trees for good growth: ample rooting media, proper aeration, and adequate moisture and nutrient supply. These soil properties are variable within a reclaimed mine soil, and individual tree species requirements are specific. Construction of reclaimed mined sites should take into account not only the mechanical processes that are required to ensure successful reclamation, but the physical and chemical conditions that result. Such considerations are crucial to the interaction between the developing soil and the planted trees.

Our study results also reinforce concerns about reclaimed mined land conditions created by new regulations based on the Surface Mining Control and Reclamation Act. Improper selection of spoil material, lack of original soil, biota, and seed pools, over-grading and compaction, and overly-competitive ground covers will all negatively influence the excellent forest productivity shown to exist on many mine soils created prior the enactment of SMCRA.

Carbon Sequestration

Carbon accreditation of forest development projects is one approach to sequestering atmospheric C, under the provision of the Kyoto protocol. The C sequestration potential of forested mined land is not well known. We characterized 14 mined and 8 adjacent, non-mined, mature forests throughout the Midwestern and Eastern coal regions to determine the C sequestration potential of mined land reclaimed prior to the passage of the Surface Mining Control and Reclamation Act of 1977. Tree, litter, and soil C data were collected from independent locations in each mined and non-mined forest. The ages of the mined stands ranged from 18 to 71 years, with an average age of 39 years, and the ages of the natural plots ranged from 36 to 87 years, with an average age of 55 years. On average, the highest amount of ecosystem C (Eco_C) 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 contained 210 Mg C ha^{-1}, which was about 62% higher than the average of all mined stands. Non-mined hardwood stands sequestered approximately 42%, 62%, and 79% more cumulative C in total tree biomass, litter, and soils, than the pine, hardwood, and mixed stands on mined land, respectively.

We developed statistically significant and biologically reasonable response surface models for ecosystem C content (Eco_C) across the spectrum of site quality and stand age. The mined land regression models explained 59%, 39%, and 36% of the variation in Eco_C in mixed, pine, and hardwood stands, respectively. Site quality had an exponential effect on Eco_C in pine and mixed stands and these models showed an Eco_C decrease with stand age. Ecosystem C increased asymptotically with stand age in mined hardwood stands but was not affected by site quality.

We also developed a set of regression equations to describe the ecosystem, total tree, litter layer, and soil carbon content projected to rotation age 60 years for all stands on mined land; we contrasted the models for mined and non-mined hardwood stands. At rotation age, the Eco_C in non-mined hardwood stands was significantly greater on high-quality sites, SI > 22 m, and was similar for low-quality sites compared to mined hardwood stands. The overall results indicated that the higher the original forest site quality, the less likely long-term productivity was restored, and the greater the disparity between pre- and post-mining C sequestration potentials.

Husbandry Practices

The effects of a range of silvicultural treatments on the survival and growth of different tree species planted on reclaimed coal mines in the Appalachians were evaluated. Three sites were selected for study, one each in Ohio (OH), West Virginia (WV), and Virginia (VA). At each site, plots were planted with there species: (1) hybrid poplar; (2) white pine; and (3) a native hardwood mix. A gradient of silvicultural treatments designed to improve survival and growth of seedlings was applied: (1) weed control only; (2) weed control plus tillage; and (3) weed control plus tillage plus fertilization. First-year survival and growth varied among the three sites and the species planted. In Virginia, where the mine soil was developed from oxidized sandstone, survival and growth were better than at the sites in West Virginia and Ohio, where siltstone- and shale-derived mine soils dominated the sites. Hardwood survival across treatments was 80%, 85% and 50% for the sites in Virginia, West Virginia, and Ohio, while white pine survival was 27%, 41%, and 58%, and hybrid poplar survival was 37%, 41%, and 72%, respectively. Growth of hybrid poplar increased in response to increasing silvicultural inputs, while growth of white pine and the native hardwoods were generally not affected by the silvicultural treatments applied. Hybrid poplar height and diameter growth were superior to those of the other species. Height growth of this species was 127 cm during the first year in the most intensive treatment at the site in Virginia. In comparison, the greatest height growth of white pine and hardwood was 9 cm and 8 cm, respectively. Detailed measurements of above-and below-ground biomass and tissue nutrient concentrations were made on the hybrid poplar at the site in West Virginia. Hybrid poplar biomass increased from 16 g to 104 g from the least intensive to the most intensive silvicultural treatment for this site. The weed control + tillage + fertilization treatment improved the foliar nutrition compared to the other treatments. The survival and growth of hybrid poplar on these sites suggest that this species may be well-suited for reforesting reclaimed mined land in the Appalachians.

This research project showed that productive forests for wood products and carbon sequestration can be established and grown on reclaimed mined land provided that reclamation is done to accommodate the needs of the forest trees. Mined land can be as productive as forest land prior to disturbance. If reclaimed with forestry in mind, costly remedial practices can be avoided and reforestation can be accomplished as part of the mining and reclamation process. Reforestation of most post-SMCRA bond released land that had been reclaimed to grass can be done, but the needed tillage, weed control, and fertilization are costly. As carbon, bio-energy, and bio-fuel markets develop, early revenues from these products and services should increase the viability of post-SMCRA reforestation.

Publications Resulting in Whole or in Part from this Sponsored Research Project

Amichev*, B. Y., and J. A. Burger. 2006. Carbon sequestration on mined land supporting grasslands: Soil organic carbon accumulation and distribution. p. 12-29. In: R. I. Barnhiesel (ed.). Proc., 23rd Annual Meeting, American Society of Mining and Reclamation. Lexington, KY.

Casselman, C. N., T. R. Fox, J. A. Burger, A. T. Jones, and J. M. Galbraith. 2006. Effects of silvicultural treatments on survival and growth of trees planted on reclaimed mined lands in the Appalachians. *Forest Ecology and Management* 223:403-414.

Casselman, C. N., T. R. Fox, and J. A. Burger. 2006. First-year survival and growth of three species assemblages planted on reclaimed mine land as affected by three levels of silvicultural intensity. pp. 200-206. *In* Proc., 13th Biennial Southern Silvicultural Research Conference. USDA Forest Service Gen. Tech. Rep. SRS-92. Memphis, TN.

Amichev, B. Y. 2007. Biogeochemistry of carbon on disturbed forest landscapes. Ph.D. Dissertation. Virginia Polytechnic Institute and State University. 371 p.

Project Objective 1a:
Forest Soil Productivity of Mined Land in the Midwestern and Eastern Coalfield Regions

Abstract

Our goal was to determine the effects of surface mining on forest land productivity in the eastern coalfields of the USA prior to the passage of the Surface Mining Control and Reclamation Act of 1977 (SMCRA), and to determine the extent to which selected mine soil properties influenced forest productivity. The site productivity of 14 mined and 8 non-mined sites in the eastern and midwestern coalfields were compared. Results show that site productivity of non-mined sites and 12 of the 14 mined sites was similar. Sites with low productivity were shallow, had high coarse fragment contents, and had lower fertility. Regression analysis identified five influential soil properties affecting site quality, which included soil profile base saturation, total coarse fragments, total available water, C horizon total porosity, and soil profile electrical conductivity. These five properties explained 52% of the variation in tree growth. Forests on most pre-law mined sites were just as productive as the forests on unmined adjacent sites and can be used as a benchmark to assess the impacts of current reclamation on mine soil quality and forest productivity.

Introduction

Surface mining has been disturbing land, forests, and waterways of the midwestern and eastern United States for over a century. Since the implementation of the Surface Mining Control and Reclamation Act in 1978, U.S. Office of Surface Mining statistics show that 500,000 ha have been disturbed by mining in the East (OSM, 1999). We estimate that an equivalent amount was disturbed prior to 1978, although the exact amount is not known. Prior to federal regulations controlling mining, the drastic nature of mining disturbance prompted some mine operators, landowners, and surrounding communities to reclaim mined areas (DenUyl, 1955), and several states with mining activity enacted regulations to control the mining process and minimize adverse effects (Davidson, 1981; Sandusky, 1980). In the midwestern and eastern states, many sites were reclaimed with trees to control erosion and reduce sedimentation. However, there was little or no expectation for commercial wood production, and the productivity of these mined lands is still largely unknown. Most mined sites planted to trees developed good vegetative cover for erosion control, but many other environmental problems remained, including degraded water quality, toxic spoils, uneven landscapes, acid drainage, highwalls, and subsidence.

The Surface Mining Control and Reclamation Act (SMCRA) was enacted in 1977 to address human safety, land productivity, and environmental problems that occurred during mining and reclamation. However, in the process of attaining these goals, reforestation disincentives were created because the reclaimed landscape is difficult to plant to trees and it is commonly unproductive for forestry (Burger, 1999). Post-law emphasis was placed on water quality and erosion control (Boyce, 1999) at the expense of site productivity, reforestation, carbon sequestration, and productive land uses. In many cases, reclamation in the Appalachian region results in mine soils that are alkaline, highly compacted, and covered with competitive grasses, which makes it difficult to re-establish forests and causes them to grow poorly (Burger, 1999). Nonetheless, the Code of Federal Regulations, 30, Mineral Resources (1997) interpreting

SMCRA requires that states restore disturbed land to conditions capable of supporting the uses which they supported prior to mining (section 715.13(a)).

An example of the mine soil productivity problem was documented by Torbert et al. (2000), who reported 11-year results of a test planting of three pine species on a pre-law mined site and a post-law mined site. They established their study during the transition period when the new law was first implemented. Trees planted on the pre-law mined site were planted on the flat bench that remained after contour coal extraction, while the post-law mined site was reclaimed to its "approximate original contour" required by the new law. The height and diameter growth of all three pine species (loblolly (*Pinus taeda L.*), Virginia (*Pinus virginiana Mill.*), and white (*Pinus strobus L.*)) was greater on the pre-law mined sites than on the post-law mined sites. The heights on the pre-law mined sites averaged 7, 5.6, and 3.7 m, while the heights on the post-law mined site averaged 6.7, 5.3, and 3.1 m for loblolly, Virginia, and white pine, respectively. The diameter growth on the pre-law mined site averaged 11.2, 9.7, and 5.3 cm, while on the post-law mined site the diameters averaged 8.9, 7.4, and 3.6 cm. Projecting these growth rates to a harvest age of 20 yr indicates that stumpage value on the pre-law site will be approximately twice that of the post-law site.

Unlike many other agricultural crops, there is no productivity standard in the regulations for forestland; a minimum number of trees surviving for the 5-yr bond period is the only performance standard associated with the tree component of forestland uses. Therefore, forest land productivity is commonly degraded in the process of mining and reclamation (Burger, 1999). Mine operators choose rock overburden material for the surface that supports vigorous herbaceous ground covers used for temporary erosion control. Research in the Appalachian coalfield region has shown that the type of overburden suitable for the temporary ground cover is not usually the best choice for long-term forest uses (Torbert, 1995). On midwestern sites, where topsoil is usually recovered and replaced, excessive grading compacts the C horizon and topsoil creating conditions unfavorable for tree establishment and growth (Pope, 1989).

In order to satisfy the intent of SMCRA, land reclaimed for forestry should logically meet a minimum productivity standard similar to that required for other crops; however, little is known about mine soil quality requirements for trees. Pre-law mined sites are growing forests in the midwestern and the eastern coalfields over a wide environmental gradient (Burger et al., 1998; Andrews, 1992; Plass, 1982). Productivity comparisons between pre-law mined sites and non-mined forest sites should show the extent to which mined sites can be reclaimed to pre-mined productivity levels. Furthermore, characterization of mined sites growing productive forests should provide insight into soil and site conditions needed for reclaiming mined land for forestry uses. Therefore, the objectives of this study were to: (1) compare the site productivity of a range of surface mined sites to adjacent non-mined forests; and (2) determine soil and site properties that influence tree growth and long-term productivity on reforested mined sites.

Methods

Fourteen forest sites across seven states, each with an average size of 2.5 ha of uniform and contiguous forest cover, were located on reclaimed mined lands in the midwestern and Appalachian coal fields (Fig. 1-1). The 14 sites ranged from 20 to 55 years old. The canopy layer species ranged from pure hardwood and conifer stands to mixed stands (Table 1-1). These sites also covered a broad spectrum of spoil types. The measurement sites were chosen to represent a cross-section of site, soil, and stand conditions. Adjacent, non-mined, native forest sites were also located and measured. These undisturbed control sites represented land and forest conditions similar to those present on the mined sites before they were disturbed. A study of pre-mined topography and landscape using maps and onsite field surveys was used to confirm that site features and soil types were similar and comparable. All non-mined sites were mature, well-stocked, native forest stands, but all had been harvested to some degree at some point in their history. After the boundaries of each study site were established, a 20 x 20 m grid was superimposed across the site. Attempts were made to place grid lines perpendicular to spoil banks on open-pit mined sites where more than one spoil bank existed to ensure that the sites' microtopography was taken into account. A 20-m buffer strip was maintained on all edges of each forest site. Sampling was based on the intersections of the grid (Fig. 1-2). Field data collection took place between May and August 1999, with the exception of two sites that were measured in August 1998.

Site productivity for forest growth is a function of soil, geology, topography and climate. Site productivity is commonly estimated using site index (SI), the average height of co-dominant canopy trees at a selected age (e.g., age 50) (Carmean, 1975). White oak was used as the indicator species and was always measured when present. To compare across sites growing different species, SI was standardized to white oak SI (Wenger, 1984). This site productivity estimate reflects the influence of soil and site characteristics while removing the effect of differences in tree species' growth patterns, tree age and stocking levels. Sampled trees were healthy, intermediately shade-tolerant, in dominant or co-dominant positions in the canopy, and had been in free-to-grow positions for most of their lives. On each of four randomly-chosen plots, tree height and total age were measured on one tree of the three main species in the canopy layer. Regional site index curves were used to estimate their height at age 50 (Carmean et al., 1989). To make direct comparisons between mined sites and non-mined sites, site index estimates for each species were converted to a site index for white oak (base age 50 yr) using Doolittle's (1958) species comparison curves. Site index relationships among species that naturally grow together in the same stand are well-correlated and their relationships are well-documented (Carmean, 1975). White oak was used as the indicator species because it is a common species throughout the region; it is equally productive across the region; it is commonly planted on reclaimed sites; and it is an important commercial species.

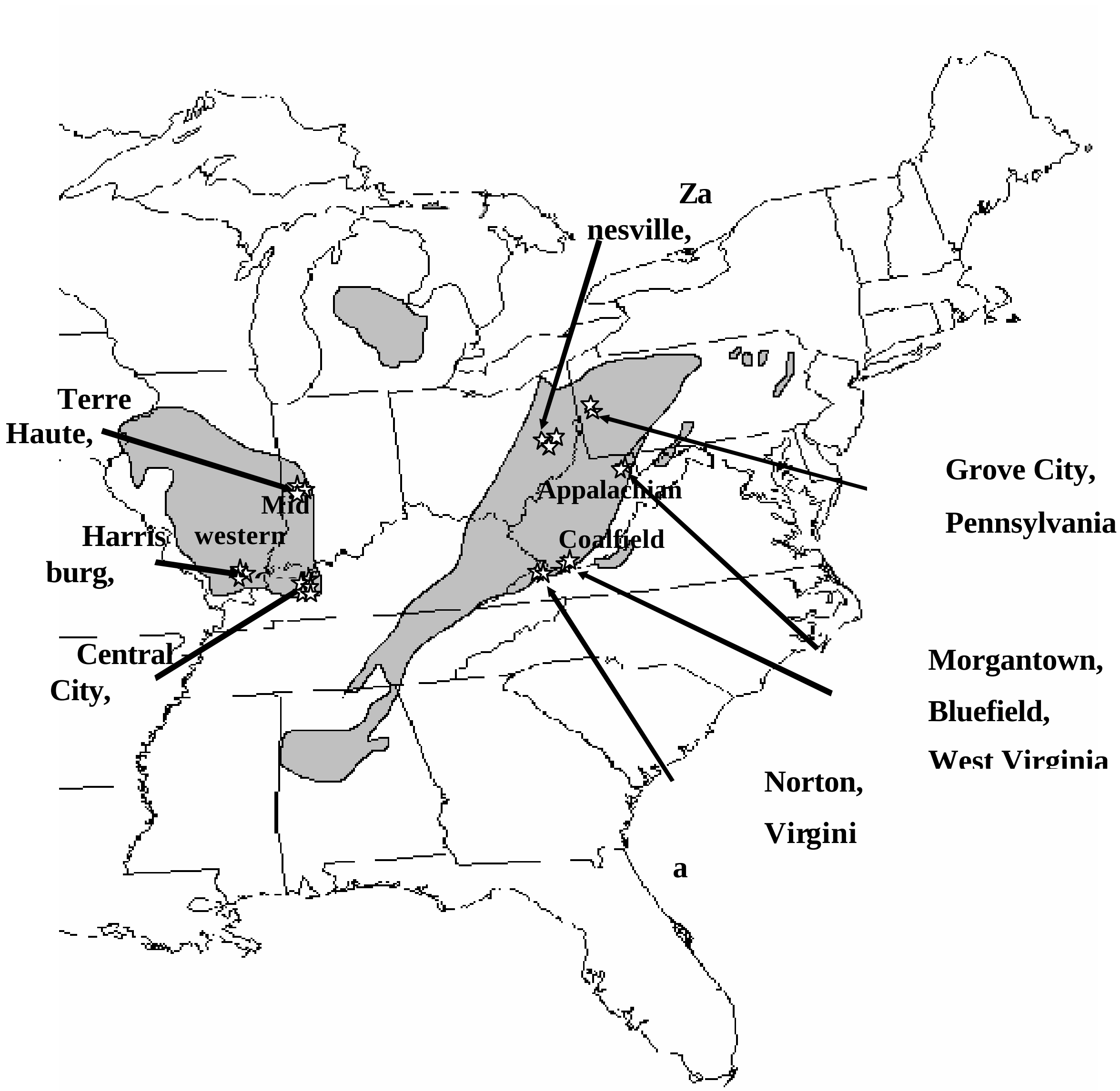

Figure 1-1. General location of study sites in the Midwestern and Appalachian coalfields.

Table 1-1. Location, description, age, and site quality of study sites located in the Midwestern and Appalachian coalfields.

State (County)	Site Name	Soil/Mine Spoil§	Regeneration History	Year Planted	SI_{50} (White Oak)† Meters (S.D.)
Illinois (Saline)	Non-mined	Aeric Haplaquepts	2nd-3rd generation bottomland hardwood site (*Q. coccinea, A. rubrum*)	Multi-aged	27.7 (4.2)
	IL-1	Open pit-mined, cast overburden	Planted to *Q. alba* and *R. pseudoacacia*. Partially underplanted with *L. tulipifera*. *R. pseudoacacia* is no longer present.	*Q. alba* 1938 *L. tulipifera* 1953	28.0 (1.3)
	IL-2	Open pit-mined, cast overburden, leveled with dragline	Planted to *P. deltoides*	1956	24.2 (0.5)
Indiana (Vigo)	Non-mined	Typic Hapludalfs	2nd-3rd generation upland *Quercus* spp., *L. tulipifera* site	Multi-aged	25.5 (4.6)
	IN-1	Open pit-mined, cast overburden	Planted to *P. rigida*	1944	23.4 (2.2)
	IN-2	Open pit-mined, cast overburden	Planted to *P. rigida*. *P. rigida* has been unsuccessful, allowing for invasion of secondary succession hardwoods and shrubs.	1949	24.2 (4.2)
Kentucky (Ohio, Muhlenberg)	Non-mined	Ultic Hapludalfs	2nd-3rd generation upland mixed *Quercus* spp. site	Multi-aged	23.8 (1.7)
	KY-1	Open pit-mined, cast overburden, top-graded	Planted to *L. tulipifera, P. occidentalis, Q. rubra, P. deltoides*	1964	22.7 (0.7)
	KY-2	Open pit-mined, cast overburden, top-graded	Planted to *L. tulipifera, P. occidentalis, Q. rubra, P. deltoides, L. styraciflua, Fraxinus* spp.	1964	25.4 (1.9)
	KY-3	Open pit-mined, cast overburden	Planted to *P. strobus* with *P. taeda* patch	1959	25.0 (4.6)
	KY-4	Open pit-mined, cast overburden	Planted to *P. taeda*	1966	23.8 (1.8)
Ohio (Perry, Noble)	Non-mined	Typic Fragiudalfs, Ultic Hapludalfs	2nd-3rd generation upland mixed *Quercus* spp. site	Multi-aged	23.1 (4.0)
	OH-1	Open pit-mined, cast overburden, lightly top-graded	Planted to *Q. rubra, P. grandidentata, Fraxinus* spp., and *L. tulipifera*	1949	25.1 (2.4)
	OH-2	Open pit-mined, cast overburden	Planted to *P. occidentalis, Q. rubra, Fraxinus* spp., *L. tulipifera*	1949	26.8 (1.2)
Pennsylvania (Mercer)	Non-mined	Aeric Fragiaqualfs	2nd-3rd generation upland hardwood site (*L. tulipifera, P. serotina, Acer* spp., *Quercus* spp.)	Multi-aged	24.8 (2.0)
	PA-1	Open pit-mined, leveled by dragline	Planted to alternating rows of *P. strobus* and *P. sylvestris*	1959	20.3 (0.7)
West Virginia (Preston, Mercer)	Non-mined (N)	Typic Hapludults	2nd-3rd generation upland hardwood site (*L. tulipifera, M. acuminata, Acer* spp., *Quercus* spp.	Multi-aged	24.7 (4.1)
	WV-1	Contour-mined, graded	Planted to *P. strobus*	1961	16.8 (1.5)
	Non-mined (S)	Typic Dystrochrepts, Typic Hapludults	2nd-3rd generation Appalachian mixed *Quercus* spp. site (*Q. alba, Q. rubra, L. tulipifera, Carya* spp.)	Multi-aged	25.9 (1.1)
	WV-2	Contour-mined, partially leveled	Planted to *P. strobus*	1971	28.7 (3.4)
Virginia	Non-mined	Typic Dystrudepts	2nd-generation Appalachian cove hardwood site (*L. tulipifera, Quercus* spp., *Carya* spp.)	Multi-aged	26.9 (1.6)
	VA-1	Contour-mined, leveled	Planted to *P. strobus*	1977	25.2 (1.6)

†SI50 (white oak): Site quality is estimated with site index (SI), the average height of a white oak canopy at age 50.

§All mine soils were classified as Udorthents.

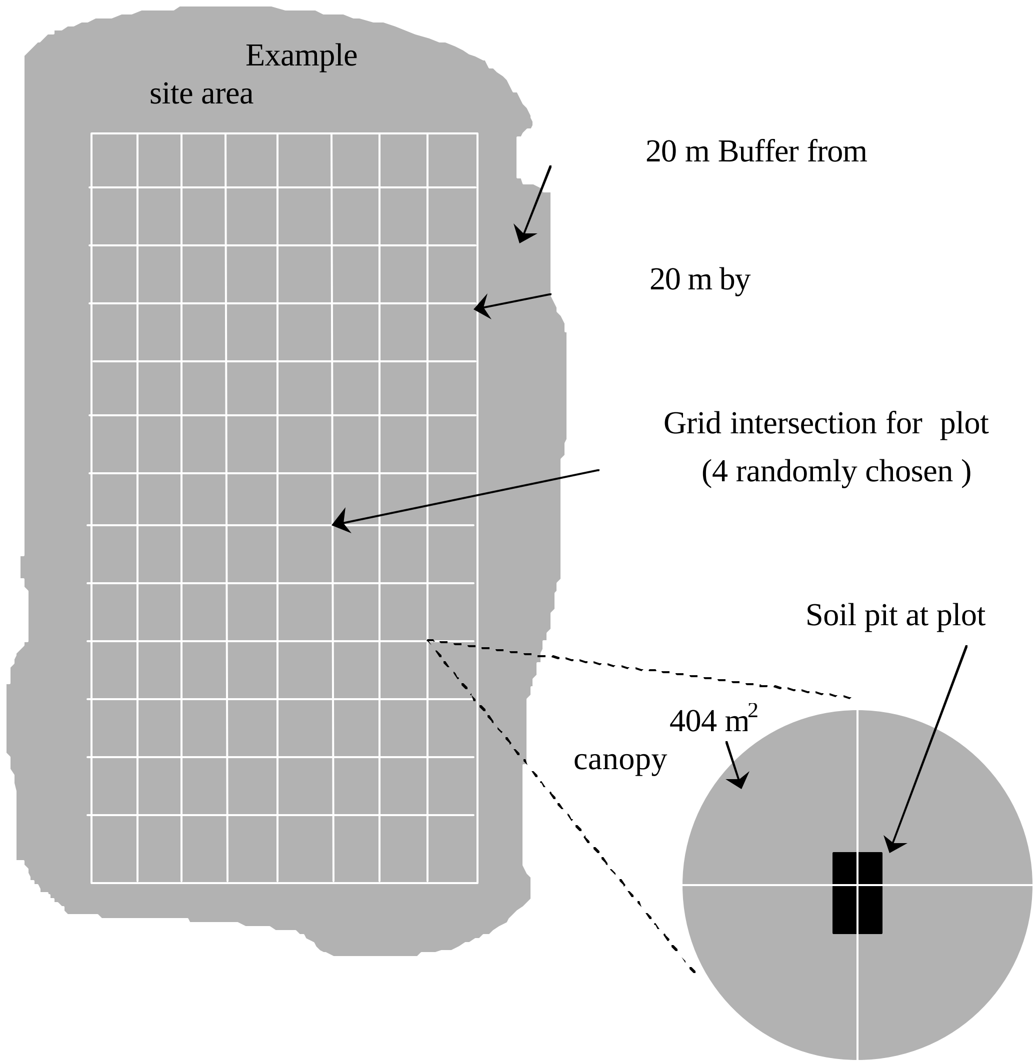

Figure 1-2: Typical site and plot layout used at each study site.

Soil pits were dug at four randomly-located plot centers on each site (Fig. 1-2). Pits were described using standard soil description techniques. Loose samples and duplicate bulk density samples were collected from each soil horizon of the profiles of both mined and non-mined sites. All minesoils were Entisols with AC profiles. The non-mined soils were Inceptisols, Ultisols, and Alfisols (Table 1-1). Soil properties were analyzed by horizon and profile average values were weighted by horizon depth. Soil samples were air-dried, sieved (2 mm), and weighed to determine coarse fragment content. Particle size was determined using the hydrometer method (Gee and Bauder, 1986). Bulk density and porosity, also corrected for coarse fragment content, were determined using soil cores. Porosity was determined using a tension table with a 50-cm water column (0.005 MPa). Available water holding capacity, water held between 0.03 Mpa and 1.5 Mpa, was determined with a pressure plate apparatus (Kulte, 1986). Soil pH was determined using a 1:2 soil/water mixture (McLean, 1982). Electrical conductivity (EC) was determined by a 1:5 soil/water extract (Rhoades, 1982). Total carbon was determined using a Leco carbon analyzer, and pedogenic carbon was estimated by the Walkley-Black wet oxidation procedure (Nelson and Sommers, 1982). Exchangeable acidity was determined using the potassium chloride extraction technique (Thomas, 1982). Exchangeable cations (Ca, Mg, K, Na, Fe, Al, and Mn) were extracted with 1 *M* ammonium acetate (Thomas, 1982). Effective cation exchange capacity 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, 1982). Total nitrogen was determined by Kjeldahl digestion and analyzed with a Bran and Luebbe TRACCS 2000 spectrophotometer (Bremner and Mulvaney, 1982). Phosphorus was extracted with sodium bicarbonate (Olsen and Sommers, 1982). Phosphorus and cations were determined using a Jarrell-Ash ICAP-9000 spectrophotometer.

Differences in site index between non-mined and mined study sites were tested using t-tests. Regression analysis was used to determine the effects of mine soil properties on site productivity (SAS Institute, 2000). Soils on each site were thoroughly characterized both in the field and laboratory by measuring 37 properties that we hypothesized influenced tree growth (Table 1-2). Values of soil properties known to be non-linear in their relationship with site index were transformed for linear regression analysis, and soil variables expressed as ratios or percent were transformed using the arcsine function. The data set was then analyzed for multicollinearity, and soil properties high in multicollinearity were removed (filtered).

The filtering process involved iterative steps using SAS software (SAS Institute, 1999). Each step involved examination of information from the variance-decomposition proportions (proportions of variation in SAS) and variance inflation factors (VIFs). The variance decomposition proportions (VDP) calculated the proportion of variance inflation contributed by each variable. A high VDP indicated that multicollinearity existed between variables. Those variables identified by the VDP were then cross-checked with the variance inflation factors which measure the combined effect of all the variables on the variance of one of the variables (Montgomery et al., 2001). More than one high VIF reinforces the presence of multicollinearity. For each step, the variable that contributed most to the multicollinearity was eliminated from the model. The filtering step was then repeated, removing one variable at each step until multicollinearity among variables was removed.

Table 1-2. Soil properties of the full profile, A horizon, and C horizon measured on all study sites. Those properties shown in bold type were used as independent variables in regressions with forest site index. Those properties shown in regular type were screened out of the variable list due to their multicollinearity with other soil variables.

Profile Properties	A Horizon Properties	C Horizon Properties
Physical Properties:		
Total depth, cm	**Depth, cm**	**Silt + clay content, %**
Coarse fragments, %	**Silt + clay content, %**	Fine earth bulk density, Mg m^{-3}
Slope, %	**Fine earth bulk density, Mg m^{-3}**	**Total porosity, %**
Rooting volume, m^3 ha^{-1}	Total porosity, %	**Capillary porosity, %**
Available water, cm	**Capillary porosity, %**	
Chemical Properties:		
EC, uS cm^{-1}	**pH**	**pH**
Exchangeable acidity, $cmol^+$ kg^{-1}		**Olsen P, mg kg^{-1}**
Walkley-Black organic C, %		**Total N, mg kg^{-1}**
Olsen P, mg kg^{-1}		Exchangeable Mg, mg kg^{-1}
Total N, mg kg^{-1}		Exchangeable Al, mg kg^{-1}
Exchangeable Mg, mg kg^{-1}		Exchangeable K, mg kg^{-1}
Exchangeable Al, mg kg^{-1}		Exchangeable Ca, mg kg^{-1}
Exchangeable K, mg kg^{-1}		**Exchangeable Mn, mg kg^{-1}**
Exchangeable Ca, mg kg^{-1}		**Exchangeable Fe, mg kg^{-1}**
Exchangeable Mn, mg kg^{-1}		
Exchangeable Fe, mg kg^{-1}		
CEC, $cmol^+$ kg^{-1}		
Base saturation, %		

Three regression model selection procedures (R^2, Stepwise, MaxR) were used on the filtered mine soil dataset to determine the combination of mine soil properties that accounted for the variation in site index on the mined sites. The use of multiple model selection procedures allowed of the data using each procedure's strengths, the examination of multiple models, and better knowledge of variable relationships. After regression analysis, the best model was selected based on criteria that included minimizing MSE and maximizing individual variable significance, adjusted R^2, R^2, and biological significance. Results from statistical tests termed "different" in this paper have a significance level of $p \leq 0.10$.

Results and Discussion

Mined Site Characterization and Productivity

Mined sites included in this study were located in seven states, were mined with different techniques, and were planted with a variety of tree species (Table 1-1). On the flat terrain of the midwest, most mining was done in open pits and overburden was cast in piles and left unleveled. However, IL-2 was leveled with a dragline; KY-1 and KY-2 were top graded, meaning that the tops of cast piles were struck off with a dozer. Other midwestern sites (IL-1, IN-1, IN-2, KY-3, KY-4) were left unleveled. The eastern sites were a mixture of open-pit and contour mined sites, with a highwall remaining on one side adjacent to a relatively flat, narrow bench with overburden piled on the outslope. OH-1, OH-2, and PA-1 were open pit-mined. OH-1 was top graded and OH-2 was left unleveled. PA-1 was open pit-mined and leveled by a dragline. The others were contour-mined (WV-1, WV-2, VA-1). Half of the midwestern sites were planted with hardwood species, while pines were commonly used on eastern sites. The oldest sites were planted in 1938 and the most recent in 1977.

Mined site productivity (site index) ranged between 16 and 27 m across both coalfields, averaging 24 m (Table 1-1). On midwestern mined sites, site index ranged from 23 to 28 m (24.6 m average). Site index on eastern mined sites varied between 17 and 29 m (23.8 m average). Non-mined site index was less variable, ranging from 23 to 28 m with an average of 25 m. The greater variation in SI on mined versus non-mined sites was probably due to the mix of mine spoil types and properties represented on our study sites.

Average site productivity of mined sites in the midwestern region was the same as that of their non-mined counterparts (Fig. 1-3). Post-mining site index across all midwestern sites varied within a range of ±10%, but these differences were not significant. Most importantly, this result indicates that pre-law mined sites in the midwestern coalfields are at least as productive after mining as before. Site productivity of the eastern sites was lower on two out of six. Though statistically similar, the site index of the mined Ohio sites appeared to be 10 to 15% higher than the non-mined condition. Both of the mined sites PA-1 and WV-1 clearly had lower site indices than the non-mined condition. The 32% difference in productivity on WV-1 was attributed to a soil with high coarse fragments and low base saturation. Average coarse fragment content was 82%, and the base saturation was 36%, the lowest base saturation level measured throughout the mined sites. Similar soil environments have been identified on poor-quality sites throughout the eastern coalfield region (Daniels and Amos, 1981, Torbert et al., 1994). Mined site PA-1 was similar to WV-1, with low base saturation and high coarse fragments. The site indices of PA-1 and WV-1 were 18% and 32% lower than those of their respective non-mined sites (WV-C1, PA-C, Fig. 1-3).

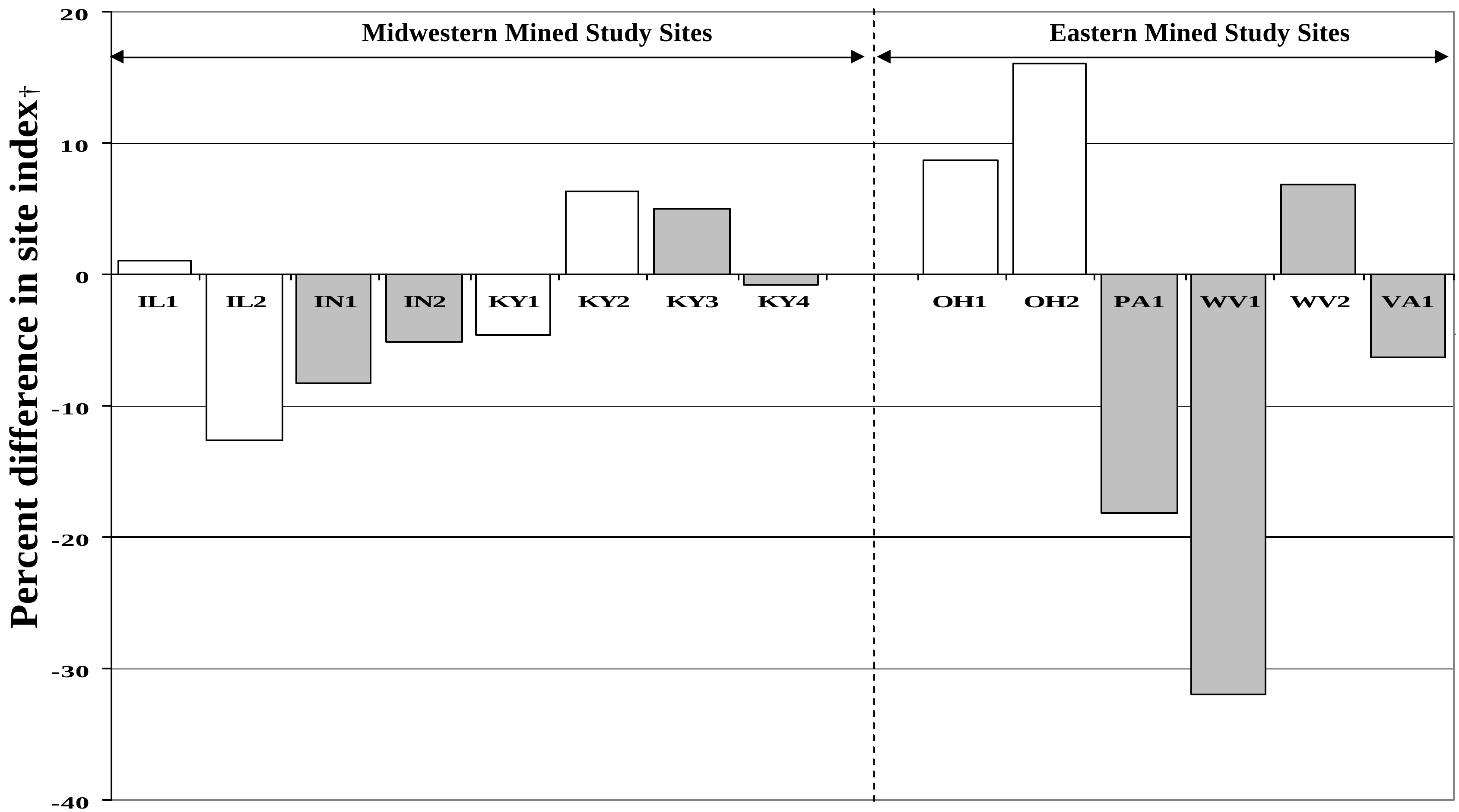

Figure 1-3: Relative productivity of non-mined (zero baseline) and mined sites in the midwestern and eastern coalfields.

Grey bars represent mined sites planted to pine species; open bars represent mined sites planted to hardwood species.

* Different between mined and non-mined sites ($p \leq 0.10$).

† Site index is the height of a white oak canopy at age 50.

Mine Soil Properties and Site Productivity

Regression Model

Regression analyses were used to determine which mine soil properties most influenced forest productivity. One site property and 36 soil properties were included in the analysis (Table 1-2). The filtering process for multicollinearity, as described in the Methods section, identified soil properties causing multicollinearity in the C horizon and the total profile. Due to the young age of these soils and poor horizon development, the C horizon typically dominated the soil profiles on each site, resulting in few differences among nutrient values in the C horizon and the total profile. Two-thirds of the multicollinearity analysis involved removal of soil variables related to fertility. For example, exchangeable Ca in the C horizon, exchangeable Ca in the profile, and exchangeable Mg in the C horizon were found to be related. Exchangeable Mg contributed most to the multicollinearity so it was removed. In another example, bulk density and total porosity in the C horizon were correlated; bulk density in the C horizon was removed. After filtering for multicollinearity, the data set contained 22 independent soil and site variables, 12 physical properties, and 10 chemical properties (depicted in bold letters in Table 1-2) that were subsequently used as independent variables in the regression analyses. Each of the three regression procedures (R^2, Stepwise, MaxR) selected the same best model, which included the same five soil variables. All five variables had partial tests that were significant in the model at $p \leq 0.10$. The final model had one of the highest adjusted R^2, one of the lowest MSE of models that met the partial test criteria, and all variables were biologically relevant to developing forests on surface mines. The model is:

$$\mathbf{SI\ WO_{50} = 18.5 + 8.0\ (ln\ (arcsine(BS))) \cdot 5.1\ (ln\ (arcsine(CF))) + 16.4\ (ln\ (AWHC)) + 34.2\ (arcsine\ (TP_c)) \cdot 4.6\ (ln(EC))} \quad \textbf{(Eq. 1)}$$

where: SI WO_{50} is site index for white oak at age 50; ln(arcsine(BS)) is the natural log of the arcsine transformed base saturation; ln(arcsine (CF)) is the natural log of the arcsine transformed coarse fragment content; ln(AWHC) is the natural log of available water holding capacity; arcsine(TPC) is the arcsine transformed C horizon total porosity; and ln(EC) is the natural log of electrical conductivity.

The final R^2 for the model was 0.52. The model contained four variables representing the entire mine soil profile (BS, CF, AWHC, EC) and one variable associated with the C horizon (TP_C); three were physical properties (CF, AWHC, TP_C) and two were chemical properties (BS, EC). All soil properties included in the model can be measured using standard procedures.

Scattergrams of site index as a function of these five soil properties are presented in Fig. 1-4. For four of the five variables, the natural relationships were best approximated with a natural log or asymptotic function (Fig. 1-4F). For example, the natural relationship between site productivity and base saturation (Fig. 1-4A) increases rapidly at low base concentrations, but levels off as base concentrations increase and tree nutrient requirements are met. Site index as a function of AWHC is similar (Fig. 1-4C). Conversely, site index is high at low levels of coarse fragment content (Fig. 1-4B) and soluble salt concentrations (Fig. 1-4C). The transformed soil properties represent the linearized form of the natural log transformations (Fig. 1-4A, B, C, E), which were used in the linear regression analysis. Site index increased proportionally across the range of C horizon total porosity (Fig. 1-4D); therefore, the simple linear relationship was used in regression analysis for this soil property.

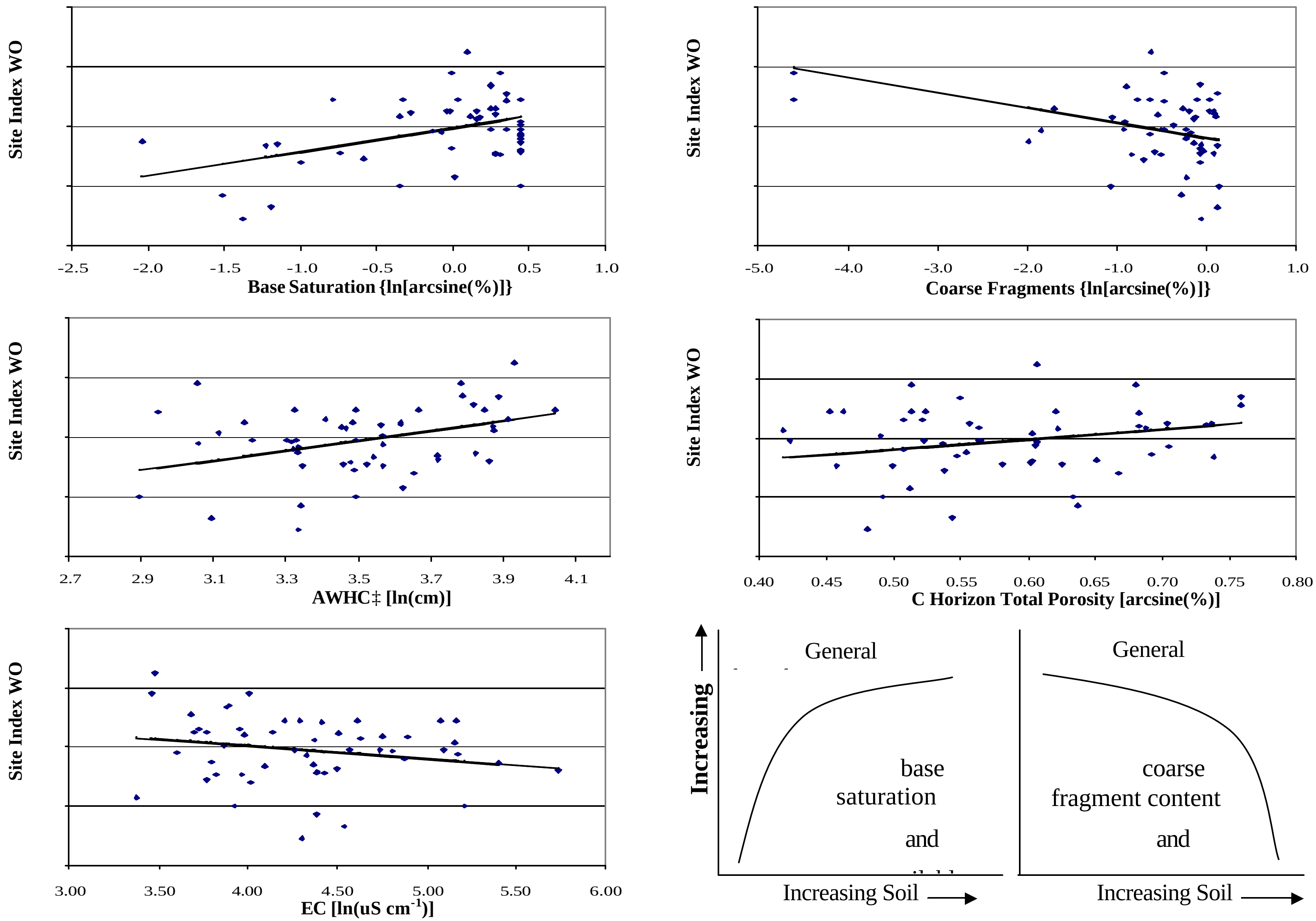

Figure 1-4. Distribution of site index as a function of five selected mine soil properties across 14 mined study sites. Fig. 1-4F shows the non-transformed, general functions for the soil properties that are non-linear with site index.

†Regression line depicts predicted site index based on observed values of each soil variable while others held constant at their average.

‡AWHC represents cm of water held between field capacity and wilting point for the total profile depth on each site.

Standardized Coefficients

Standardized coefficients were determined for the five soil variables in the regression model. The coefficients show the relative influence of each variable on site index (Table 1-3). One standard deviation increase in the independent variable changes the dependent variable, site index, by the dependent variable's standard deviation times the standardized coefficient of the independent variable. For example, an increase in one standard deviation of profile base saturation (s.d. = 26%) results in a 1.46 m increase in site index. Profile coarse fragments (s.d. = 27%) decreased site index by 1.44 m, while profile available water (s.d. = 9cm) increased site index by 1.33 m. C horizon total porosity (s.d. = 7%) increased site index by 0.92 m. Profile electrical conductivity (s.d. = 0.54dS m^{-1}) reduced site index by 0.74 m.

Table 1-3. Standardized coefficients and statistics for the independent variables in Equation 1.

Variable	Standardized Coefficient	Change in Site Index (m)	Partial Tests	Model R^2
ln ((arcsin) base saturation)	0.43123	1.46	.0002	0.1742
ln ((arcsin) coarse fragments)	-0.42674	-1.44	.0004	0.2746
ln (available water holding capacity)	0.39222	1.33	.0007	0.3509
Arcsine (C horizon total porosity)	0.27111	0.92	.0141	0.4105
ln (electrical conductivity)	-0.21972	-0.74	.0406	0.5183

Interaction effects of multiple soil properties are shown in Fig. 1-5, plots of site index as a function of the master variable AWHC and the other four significant soil properties (BS, CF, TP_c, EC). Each plot shows the predicted site index across the range of AWHC for five levels of a second soil property while all other variables were kept at their mean value. Across the range of AWHC (18 to 57 cm) and BS (13 to 100%), predicted site index ranged from approximately 17 to 28 m. At 35cm AWHC, site index varied 5 m across the range of BS. There was a proportional increase in site index with increasing base saturation. In other words, a 20% increase in BS resulted in approximately a 1-m increase in site index across the range of BS. This proportional relationship also occurred for 10% increases in TP_C, which resulted in 1.5 m increases in site index. Across the range of AWHC and TP_C (41 to 67%), site index increased 11 m. At constant AWHC, a 4.5-m increase in site index resulted from an increase in TP_C from 30 to 70%.

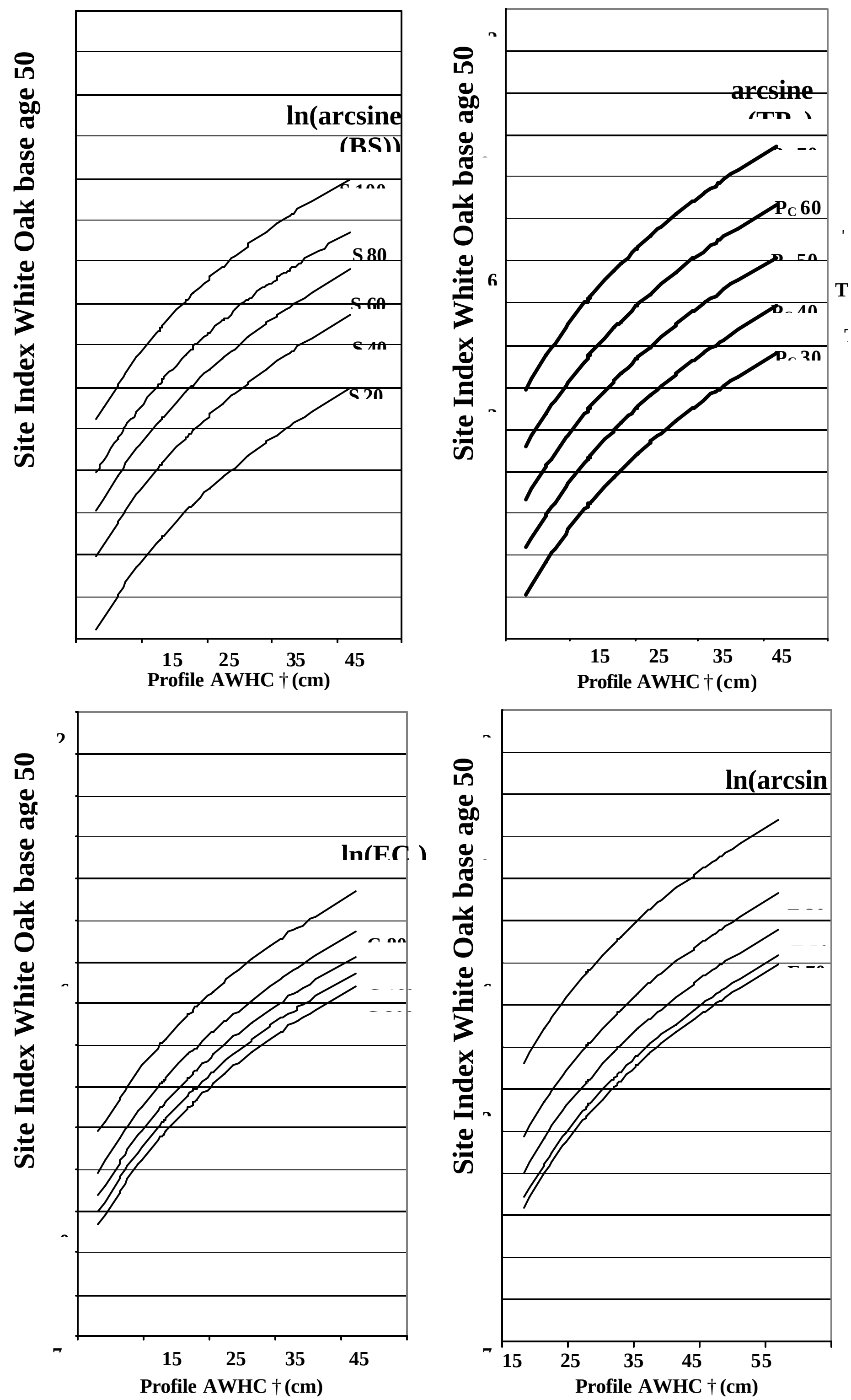

Figure 1-5. Estimated site index over a range of profile AWHC as a function of mined site base saturation, C horizon total porosity, electrical conductivity, and coarse fragments.

† AWHC represents cm of water held between field capacity and wilting point for the total profile depth on each site.

The combined influence of AWHC and EC resulted in an 8-m change in site index (Fig. 1-5). At constant AWHC, a fivefold decrease in EC resulted in a 3-m increase in site index. The relationship between AWHC and EC was not linear; a decrease in EC from 200 to 160 uS cm^{-1} resulted in a smaller increase in site index (0.5 m) than a decrease from 80 to 40 uS cm^{-1} (1.5 m). This was also the case with CF. At constant AWHC, a decrease in CF from 30 to 10% resulted in a greater increase in site index than a decrease from 90 to 70%. Over the range of AWHC, a decrease in CF from 90 to 10% resulted in a 9.5-m increase in site index. For both EC and CF, changes at higher concentrations resulted in small changes in site index. This suggests that within the range of AWHC, higher levels of EC and CF are above an acceptable threshold for forest site productivity.

Mine Soil Properties Influencing Tree Growth

Base Saturation

Base saturation was the single most important soil variable influencing mined site productivity. Across all mined sites, site productivity increased as base saturation increased. Base saturation ranged from 13 to 100% and in most cases was higher than that of the non-mined sites (Table 1-4). The distribution of base saturation was skewed towards high levels (Fig. 1-4), between 80 and 100%, with the highest levels found, on average, on sites in the Midwest (i.e., IL-1, IL-2, IN-1, IN-2, and KY-1) (Table 1-4). Base saturation of these midwestern mine soils were the least variable. These sites contained significant amounts of calcium and magnesium in relation to other cations such as aluminum and hydrogen. High base saturation levels (>50%) represent adequate base cation availability and a low amount of exchangeable acidity. Unlike typical forest soils of the region, mine soils are commonly composed of freshly-weathering material containing higher amounts of permanent charge than the pH-dependent charge associated with organic matter and highly-weathered minerals. After several decades, mine soils with base saturation levels approaching 100% indicate that buffering capacity is still strong, likely due to large amounts of Ca and Mg weathering directly from carbonates. Czapowskyj (1978) found that calcium present in some Pennsylvania mine soils accounted for approximately 80% of base saturation. Despite a broad range in pH from 3.2 to 7.9, this soil property was not significantly related to SI. Native oaks are commonly found across this pH range and are fairly tolerant of pH extremes. A positive SI correlation with base saturation is likely due to base availability rather than associated neutral pH levels.

Mined site base saturation and cation nutrition are highly dependent on the parent material. In northern Pennsylvania, Czapowskyi (1978) identified a base saturation range of 45 to 85% depending on the parent material. This spread in base saturation contributed to a wide range in poplar (*Populus* spp.) height growth (3 to 14 m). The base saturation of our other mined sites varied widely among plots, with standard deviations as high as 38% (Table 1-4). WV-1, the site with the lowest forest productivity, had three plots with base saturation less than 30%. Site PA-1 had an average site index estimate of 23 m and an average base saturation of 51%, but with plots as low as 30%. Other research on Pennsylvania mine soils reported base saturation as low as 10% (Pedersen et al., 1978). Cummins et al. (1965) reported that greater than 70% of eastern Kentucky spoils sampled were below 60% base saturation, which would adversely affect agronomic plant growth on these sites.

Table 1-4. Values of selected soil properties for 22 soils in the Midwestern and Appalachian coalfields.

Site	Base Saturation % (s.d.)	Coarse Fragments % (s.d.)	Available Water Holding Capacity† cm (s.d.)	C Horizon Total Porosity % (s.d.)	Electrical Conductivity uS cm^{-1} (s.d.)
IL-C‡	**67 (17)**	**4 (1)**	**17 (3)**	**57 (2)**	**86.8 (14.2)**
IL-1	94 (6)	14 (19)	32 (12)	50 (2)	57.1 (11.0)
IL-2	99 (2)	47 (10)	27 (6)	55 (2)	152.4 (37.7)
IN-C	**68 (23)**	**6 (5)**	**25 (5)**	**47 (3)**	**47.8 (4.0)**
IN-1	95 (8)	59 (13)	31 (6)	48 (4)	47.4 (4.0)
IN-2	100 (0)	50 (16)	33 (5)	44 (3)	158.4 (30.9)
KY-C	**39 (26)**	**13 (9)**	**19 (2)**	**52 (2)**	**52.2 (7.5)**
KY-1	99 (1)	68 (12)	35 (7)	56 (6)	116.8 (63.4)
KY-2	83 (19)	74 (12)	29 (8)	65 (2)	95.6 (25.9)
KY-3	83 (12)	83 (6)	49 (6)	51 (8)	88.0 (21.0)
KY-4	63 (30)	78 (8)	36 (12)	52 (3)	60.2 (24.5)
OH-C	**66 (22)**	**31 (2)**	**14 (4)**	**52 (5)**	**50.5 (9.6)**
OH-1	79 (25)	53 (33)	39 (10)	58 (1)	158.7 (100.6)
OH-2	94 (8)	78 (16)	41 (5)	67 (3)	51.2 (9.6)
WV-C1	**23 (14)**	**42 (7)**	**17 (4)**	**52 (3)**	**57.4 (17.7)**
WV-1	36 (20)	82 (11)	24 (5)	54 (6)	74.4 (17.9)
WV-C2	**16 (9)**	**41 (28)**	**15 (2)**	**64 (6)**	**50.0 (15.3)**
WV-2	70 (38)	52 (30)	39 (11)	59 (6)	37.2 (6.2)
PA-C	**16 (12)**	**32 (1)**	**12 (3)**	**54 (4)**	**50.7 (9.6)**
PA-1	51 (25)	72 (18)	36 (3)	57 (9)	47.1 (13.8)
VA-C	**36 (12)**	**41 (20)**	**15 (5)**	**58 (3)**	**40.7 (4.8)**
VA-1	----- §	----- §	----- §	58 (7)	58.6 (28.3)

† AWHC represents cm of water held between field capacity and wilting point for the total profile depth on each site.

‡ **Bold** sites are non-mined control sites.

§ Data not available.

Coarse Fragments

The second most important variable in the regression model was total profile coarse fragments. Coarse fragments negatively impacted site index, which decreased as coarse fragments increased. Average coarse fragments on mined sites ranged from 14% to 83% (Table 1-4). A mid-range gap in data shows that the relationship with SI may be unduly influenced by several sites with low values (Fig. 1-4B). Ten of the 14 mined sites had coarse fragments greater than 50%. The coarse fragment content of all non-mined sites was lower than 50%. Mined sites in this study consisted of cast overburden or contour bench sites high in rock content throughout the profile. Excessive amounts of coarse fragments limit the fine earth volume available for root proliferation, water-holding capacity, and long-term nutrient availability (Torbert et al., 1988; Childs and Flint, 1988; Thurman and Sencindiver, 1986; Lyford, 1964). Other researchers have noted similar ranges in rock content on mined sites and their impact on plant growth (Andrews et al., 1998; Torbert et al., 1988; Pedersen et al., 1978). The amount of rock present on mined sites, even after a period of 20 to 55 years, depends on rock hardness, blasting techniques, and spoil handling (Daniels and Zipper, 1997). Several researchers reported a reduction of coarse fragments with time in surface horizons where weathering processes are rapid (Daniels and Zipper, 1997; Johnson and Skousen, 1995; Haering et al., 1993). Surface horizons within this study commonly contained lower coarse fragment percentages; however, the high C horizon rock content of some of our oldest sites indicate that weathering processes are only beginning to influence the mine subsoils.

Available Water Holding Capacity

Water availability is the most important growth-promoting factor for many native forest types (Pritchett and Fischer, 1987) and is of significant importance on reclaimed mined sites (McFee et al., 1981; Czapowskyj, 1978). Profile AWHC was the third most important soil variable influencing mined site productivity (Table 1-3). As AWHC increased, site index increased. AWHC, defined in this study as the amount of water available between field capacity and wilting point for the whole profile depth (cm), ranged from 18 to 57 cm across all sites (Table 1-4).

Mined sites can have poor water retention resulting from high coarse fragment content, lack of fine earth, and poor soil structure, which allow water to drain quickly from the soil profile (Thurman and Sencindiver, 1986, Pedersen et al., 1978). However, AWHC was higher on all mined sites compared to adjacent non-mined sites. On average, the mined sites were 48% higher. Thurman and Sencindiver (1986) also observed similar subsurface water retention on several mined sites compared to local native soils in the Appalachian region. Similarity in AWHC levels was due to deeper mine soils compared to native soils. In our case, simple relationships did exist between available water and total depth, coarse fragments, and C horizon silt and clay percentages. As total depth and C horizon silt and clay percentages increased, profile available water increased. Conversely, as coarse fragments increased, total available water decreased.

Total Porosity

Profile total porosity was the fourth most influential variable on mine soil quality. Higher total porosity resulted in higher site productivity. Total porosity of non-mined forest soils generally range from 30 to 65% (Pritchett and Fisher, 1987). Total porosity ranged from 44 to 67% across all mined sites, with most soils falling in the range of 50 to 60% (Table 1-4). Total porosity of mine soils reported in other studies ranged from 27 to 83% (Andrews et al., 1998; Johnson and Skousen, 1995; Torbert et al., 1988; Indorante et al., 1981). Total porosity levels were similar to those found in adjacent non-mined soils (Table 1-4).

Non-capillary porosity influences a soil's ability to drain and exchange gases (Brady and Weil, 1999). Capillary porosity enhances the ability of soils to retain water under levels of increasing moisture stress. Non-capillary porosity ranged from 13 to 42%, indicating that conditions for gas exchange were more than adequate (Pritchett and Fisher, 1987). Ungraded cast overburden suffers little compaction from traffic that commonly occurs on post-law mine soils (Sencindiver and Ammons, 2000). Non-capillary porosity on contour-mined sites (WV-1, WV-2, VA-1) was well above 10%, indicating that traffic from mining activity did not excessively compact the mine soils. However, the combination of high coarse fragments, high non-capilllary porosity, and excess voids could make these well-drained mine soils droughty during dry periods of the year (Amundson et al., 1986; Thurman and Sencindiver, 1986). Creating deep mine soils may be one way to offset the potential droughtiness of highly porous mine soils (Wade et al.,1985; Sencindiver and Smith, 1978).

Soluble Salts

Profile soluble salts, estimated by electrical conductivity (EC), was the least significant variable in the final model (Table 1-3). However, EC is a common mine soil variable influencing plant productivity (Andrews et al., 1998; Torbert et al., 1988; Davidson, 1986, McFee et al., 1981). High levels of soluble salts inhibit water and carbon dioxide uptake, and also inactivate enzymes affecting protein synthesis, carbon metabolism, and photophosphorylation (Taiz and Zeiger, 1991). Our regression analysis indicated a decrease in site productivity with an increase in the soluble salt concentration. Andrews et al. (1998) and Torbert et al. (1988) reported a similar trend with white pine planted on mined sites.

Torbert et al. (1988) found a significant relationship between EC and finely textured soils derived from shales and siltstones, with EC increasing with finely-textured shales. EC on their study sites in Virginia ranged from 300 to 1700 uS cm^{-1}. To minimize adverse effects of EC, they recommended placing coarse-textured, oxidized sandstone on the surface instead of finely-textured, reduced overburden. Our field descriptions also showed that the finely-textured C horizons had the highest EC. Textures of plots with the five highest EC readings had textures of silty clay and silty clay loam, while the plots with the five lowest EC values had textures of sandy loam and loam.

McFee et al. (1981) listed EC as one of the most influential soil properties on Indiana mine soils. EC levels were high enough in some black and gray shales and sandstone to retard plant growth. Soluble salt concentrations greater than 1000 to 3000 uS cm^{-1} were found to be detrimental to plant growth, reducing tree survival and crop yields (McFee et al., 1981; Cummings et al., 1965). EC on our mined sites ranged from 37 to 159 uS cm^{-1} (Table 1-4), falling well below established critical limits defined for agronomic purposes. Davidson (1986) found specific conductance important to growth and survival of three pine species and two hardwood species. These collective studies suggest that forest trees may be more sensitive to salty soils than most agronomic crops. Furthermore, the influence of total salts may be manifested through symbiotic relationships with soil biota, namely mycorrhizal fungi. However, cause and effect relationships have not been determined or studied.

Other studies using regression techniques to link measures of site productivity to mine soil and site properties included the five variables in our final model; however, other soil variables such as total depth, soil N, organic carbon, and soil P were also found to be important for normal tree growth. In our study, some of these properties had co-linear relationships with the properties in our regression model. For example, total depth was colinearly related to coarse fragment content. Total depth on reclaimed mined sites is important to tree and plant growth (Andrews et al., 1998; Wade et al., 1985; Pederson et al., 1978; Scencindiver and Smith, 1978). Mine soils deeper than native soils provide trees with more exploitable volume for nutrients, water, and physical stability (Plass, 1982; Scencindiver and Smith, 1978). Our maximum sampling depth was 1.5 m, but even on sites with depths less than 1.5 m, depth was usually limited by excessively large coarse fragments rather than bedrock or compacted layers. Soils were loose enough to allow tree roots to explore avenues around large coarse fragments.

Other soil properties such as soil nitrogen, organic carbon, and phosphorus have been reported as growth limiting on mined sites, but usually within the first 10 yr after disturbance (Andrews et al., 1998; Torbert et al., 1988; Czapowskyj, 1978; Woodmansee et al., 1978; Ashby and Baker, 1968). Organic matter and total nitrogen are good indicators of nitrogen availability in minesoils (Bendfeldt et al., 2001; Woodmansee et al., 1978). Nitrogen is commonly found limiting on mined sites soon after reclamation when little organic matter is present and nitrogen fixing organisms have yet to become established. Soil profile total-nitrogen levels in this study ranged from 1208 to 5868 kg ha^{-1}. The lower end of this range is equivalent to levels found in agricultural soils of the Piedmont region of the Southeast. The higher end of the range is equivalent to levels found in undisturbed forest soils throughout the U.S. (Pritchett and Fischer, 1979). Total nitrogen was colinearly related to organic C, so it was not included in the regression analysis.

Intimately linked with soil nitrogen content, plant-derived organic carbon is scarce on recently mined sites and re-accumulates as the forest community develops (Bendfeldt et al., 2001). This process will occur across all mined sites varying by site age, vegetation type, site productivity, soil type, and climate. In this study, organic C was not significantly related to SI as shown by the regression analyses. Over a period of 20 to 60 years, the range of forest ages included in our study, organic matter accumulated to levels commonly observed in native forest soils. Organic carbon levels ranged from 1.9 to 8%, encompassing levels of soil organic carbon reported for soils from the southeastern region of the U. S. (3 to 6%) (Brady and Weil, 1999). However, tests for organic carbon on mined sites can include portions of the geogenic carbon pool left after mining, which inflates the estimates (Thurman and Sencindiver, 1986; Indorante et al., 1981; Pedersen et al., 1978; Cummins et al., 1965). In any case, organic C had no significant influence on the variation in SI across these mined sites.

Phosphorus, usually present in adequate supply immediately after mining, has been found to decrease as soil weathering takes place (Howard et al., 1988). Andrews et al. (1998) and Torbert et al. (1988) reported P deficiencies in young white pine growing on mine soils, but extractable P was not significantly related to SI in this study. Soil P ranged from 2 to 89 kg ha^{-1} on our mined sites. A commonly accepted critical P level for agricultural crops using the sodium bicarbonate extractant is 20 kg ha^{-1} (Ministry of Agriculture, Fisheries, and Food, 1973), which indicated that 7 out of 13 mine soils may be deficient in P by this standard. The presence of mature forests on these mined sites suggests that P was adequate early on, giving the trees time to acquire and cycle much of their P internally. Other properties such as water supply may have been more limiting, masking any P deficiency.

Conclusions

The Surface Mining Control and Reclamation Act requires that reclaimed mined sites be capable of the same productive land uses that occurred prior to mining. Our study demonstrated that mined sites can be as productive or more productive as their non-mined counterparts. Trees planted on reclaimed mined sites in the midwest prior to the passage of SMCRA grew as productively as before mining. Mining in the east degraded the quality of some sites. Sites with degraded productivity had high coarse fragment contents and shallow depths, preventing trees from exploiting the soil volume to meet their nutrient and water requirements. Improper spoil selection also created growth-limiting chemical conditions such as low base saturation and high salt content.

Tree growth over decades is influenced by myriad biotic, abiotic, and management factors; but mine soil properties had a dominant effect. Soil characteristics that had the greatest effect on tree growth included, profile base saturation, profile coarse fragments, profile available water, C horizon total porosity, and profile EC; these properties explained 52% of the variation in forest site index. Productive mine sites were commonly well-drained, ungraded mixtures of weathered coarse and fine textured materials. Base saturation was commonly greater than 70%, and coarse fragments averaged 59%. Profile available water averaged 30 cm, and C horizon total porosity averaged 55%. Profile EC averaged 87 uS cm^{-1}.

The soil properties identified by this study represent soil attributes fundamentally important to trees for good growth: ample rooting media, proper aeration, and adequate moisture and nutrient supply. These soil properties are variable within a reclaimed mine soil, and individual tree species requirements are specific. Construction of reclaimed mined sites should take into account not only the mechanical processes that are required to ensure successful reclamation, but also the physical and chemical conditions that result. Such considerations are crucial to the interaction between the developing soil and the planted trees.

The results from this study also reinforce concerns about reclaimed mined land conditions created by new regulations based on the Surface Mining Control and Reclamation Act. Improper selection of spoil material, lack of original soil, biota, and seed pools, over-grading and compaction, and overly-competitive ground covers will all adversely influence the excellent forest productivity shown to exist on mine soils created prior to the enactment of SMCRA.

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Project Objectives 1b and 1c:
Carbon Sequestration Empirical Models for Forests and Soils on Mined Land in the Eastern U. S. Coalfields

Abstract

Carbon accreditation of forest development projects is one approach to sequestering atmospheric C, under the provision of the Kyoto protocol. The C sequestration potential of forested mined land is not well known. We characterized 14 mined and 8 adjacent, non-mined forests throughout the Midwestern and eastern coal regions to determined the C sequestration potential of mined land reclaimed prior to the passage of the Surface Mining Control and Reclamation Act of 1977. Tree, litter, and soil C data were collected from independent locations in each mined and non-mined forest. The age of the mined stands ranged from 18 to 71 years, with an average age of 39 years, and the age of the natural plots ranged from 36 to 87 years, with an average age of 55 years. On average, the highest amount of ecosystem C 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 contained 210 Mg C ha^{-1}, which was about 62% higher than the average of all mined stands. Non-mined hardwood stands sequestered approximately 42%, 62%, and 79% more cumulative C in total tree biomass, litter, and soils, than the pine, hardwood, and mixed stands on mined land, respectively.

We developed statistically significant and biologically reasonable response surface models for ecosystem C content (Eco_C) across the spectrum of site quality and stand age. The mined land regression models explained 59%, 39%, and 36% of the variation in Eco_C in mixed, pine, and hardwood stands, respectively. Site quality had an exponential effect on Eco_C in pine and mixed stands and these models showed an Eco_C decrease with stand age. Ecosystem C increased asymptotically with stand age in mined hardwood stands but was not affected by site quality.

We also developed a set of regression equations to describe the ecosystem, total tree, litter layer, and soil carbon content projected to rotation age 60 years for all stands on mined land; we contrasted the models for mined and non-mined hardwood stands. At rotation age, the Eco_C in non-mined hardwood stands was significantly greater on high-quality sites, SI > 22 m, and was similar for low-quality sites compared to mined hardwood stands. The overall results indicated that the higher the original forest site quality, the less likely long-term productivity was restored, and the greater the disparity between pre- and post-mining C sequestration potentials.

Introduction

Within the eastern regions of the United States, about 650,000 hectares have been surface-mined for coal (OSM, 2007). Coal mining and the use of the mined coal for power generation are major sources of CO_2 in the atmosphere. In addition to coal combustion, surface mining for coal contributes further to CO_2 emissions because it totally removes the forest vegetation. Some forest biomass is harvested, but most is typically bulldozed in piles and burned.

A compelling argument can be made for restoring forest lands mined for coal to carbon-rich forests that existed prior to mining. The new forest will absorb some of the CO_2 emitted from the coal for which the original forest was sacrificed. Making an effort to maximize the productivity of the restored forest is also worthwhile because forest carbon pools can vary fivefold within a local edaphic gradient as a function of site quality. New productive forests will enhance the site's ability to recapture the carbon contained in the original forest and some of the carbon contained in the coal that was mined beneath it.

Under the 1992 Framework Convention on Climate Change, 153 nations agreed to mitigate global climate change by controlling greenhouse gases. The governments and industries of these nations would reduce greenhouse gases by sequestering atmospheric carbon or by reducing CO2 emissions (Wright et al., 2000). Carbon accreditation of forest development projects is one approach to sequestering atmospheric carbon under the climate change agreement. Forests provide a low-cost method of carbon accreditation compatible with other environmental, economic, and social development projects (Wright et al., 2000). Forest development projects use trees to sequester carbon for long-term storage. As young forests develop, atmospheric carbon is locked into wood during growth and stored in litter layers and the soil. However, some workers suggest that sequestration of carbon in tree biomass and litter is a delaying tactic that only buys time for finding more permanent solutions for C sequestration (IPCC, 2000). In any case, the carbon sequestration potential of a forest depends on stand growth rates, the site's biological carrying capacity, stand age, and product utilization. Furthermore, carbon sequestration and storage may be increased if forests are harvested and trees are converted into wood products (Skog and Nicholson, 1998).

The eastern deciduous hardwood forest of the Midwestern and Eastern regions of the United States stores large amounts of carbon in forest biomass and forest soils. Anderson (1991) estimated average worldwide carbon levels for temperate deciduous forests at 175 Mg C ha^{-1}, with 90 Mg C ha^{-1} in the plant biomass and 85 Mg C ha^{-1} in the soil and litter layer. The average carbon content of forests in the eastern half of the U.S. was 179 Mg C ha^{-1}, including 81 Mg C ha^{-1} in the forest biomass, 9 Mg C ha^{-1} in the litter layer, and 89 Mg C ha^{-1} in the soil (derived from Turner et al., 1995). These studies provide baseline examples of total carbon content and its distribution to ecosystem components in temperate deciduous forests.

Carbon content between ecosystem components can vary by forest-specific conditions. Although litter layer C content could be significantly lower than the amounts sequestered in tree biomass and soil, the litter layer plays an important role in the biogeochemical cycle of terrestrial C. Litter layer carbon pools from studies in the eastern U.S. ranged from 4 to 14.4 Mg C ha^{-1} depending on age and forest species composition (Hoover et al., 2000; Kaczmarek et al., 1995; Van Lear et al., 1995; Vose and Swank, 1993). However, sites with higher conifer components tend to develop greater carbon pools within the litter layer. In Ohio, Vimmerstedt et al. (1989) found that hardwood litter layer carbon pools (3 Mg C ha^{-1}) were significantly lower than litter layer carbon pools under pine sites (8 Mg C 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).

Van Lear et al. (1995) found that pine litter layers on mined land in the southeast tend to reach a steady state 15 to 20 years following logging. Yanai et al. (2000) found that litter layers in logged northern hardwood forests increased with age after disturbance until about 50 to 55 years. However, over the chronosequence of their study, litter accumulation was also related to the logging practice used on each site.

Carbon is also incorporated into the soil via root turnover, litter decomposition, and turnover of meso- and micro-fauna. Post et al. (1982) reported average soil carbon levels of 79 and 60 Mg ha^{-1} for the 0-100 cm depth for dry and moist warm temperate forests, respectively. Researchers in the eastern U.S. reported carbon levels for depths from 50 cm to bedrock ranging between 36 and 130 Mg ha^{-1} (Hoover et al., 2000; Johnson et al., 1995; Kaczmarek et al., 1995). Van Lear et al. (1995) reported 37.2 Mg C ha^{-1} within the top 1 meter of a degraded natural pine site on the Piedmont of South Carolina. Due to poor agricultural farming practices, this site was severely eroded, resulting in loss of the surface horizons prior to pine stand establishment. Akala and Lal (1999) reported that 30-year-old reforested mined sites contained 51.5 Mg C ha^{-1} and 50-year-old sites contained 54.9 Mg C ha^{-1} for 0-50 cm soil depth.

In another study, Akala and Lal (2001) reported soil C sequestration rates for forest treatments on mined sites in Ohio over a 21-year period ranging from 0.7 to 2.3 Mg C ha^{-1} yr^{-1} for the 0-15 cm depth and from 0.3 to 0.4 Mg C ha^{-1} yr^{-1} for the 15-30 cm depth. The strip mine reclamation law of 1972 in Ohio made mandatory the placement of topsoil on the spoil surface before revegetation. Akala and Lal (2001) showed that topsoil application led to an increase of the carbon sequestration rate in the surface layer of forested mine soils, while the rate for the 15-30 cm depth remained unchanged for a 21-year reclamation period in Ohio.

Upon analysis of the data reported for undisturbed forests in the eastern U.S. by Turner et al. (1995) and the mined land data by Akala and Lal (2001), one could estimate the time in years until the soil carbon pool on mined land reaches that of undisturbed forests. The results of such analyses indicate that the SOC pool on forested mined land in Ohio should reach the SOC levels of undisturbed forests between 33 and 89 years following forest establishment. Certainly, performing the latter analysis could raise many questions regarding the effect of mine site quality, forest type, stand age, previous land use, to mention a few, on the recovery potential of the SOC pool on mined lands. The effects of one mined land factor could interfere with the effects of another, and as a result, erroneous inferences could be made about the C sequestration potential of mined land. Therefore, carefully designed studies are needed to compare, in a defendable and valid approach, the carbon sequestration potential of forests developed on mined land to forests developed on non-mined land of similar site quality, similar forest type, equal stand age, and similar land use history.

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 differences in carbon capture levels under varying forest and mined land conditions. Therefore, we characterized 14 mined forests 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 (1) to estimate and compare the ecosystem C content sequestered on disturbed and undisturbed forest land, (2) to determine the effects of mine soil quality and stand age on the C sequestration potential of forested mined land, and (3) to develop empirical models for C sequestration in tree biomass, litter layer, and soil of coniferous and deciduous forests developed on mined land.

Methods and Materials

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 study by Rodrigue (2001) and Rodrigue and Burger (2004) was 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 on 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 from 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. (Fig. 2-1). 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).

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 mined 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 (Fig. 2-1). 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.

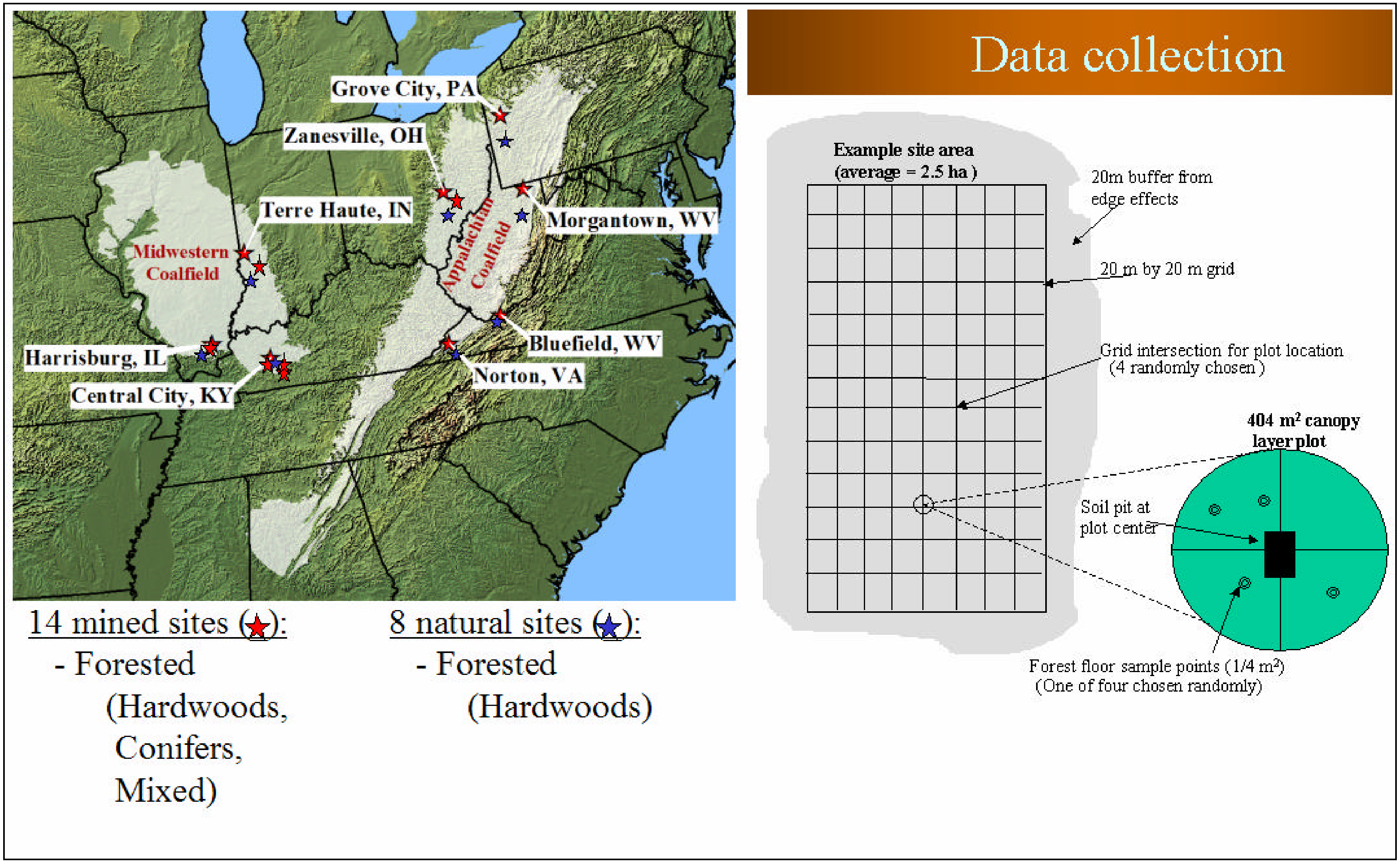

Figure 2-1. Mined and non-mined forested study site approximate locations and data collection procedures used by Rodrigue (2001) and Rodrigue and Burger (2004).

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 2-1). 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 2-1).

Rodrigue (2001) measured tree diameter, height and litter layer biomass on each plot using the data collection scheme shown in Figure 2-1. 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-m^2 sample. Bulked 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 (Fig. 2-1) 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 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 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 fragments. Soils on the mined sites were classified as Udorthents (Rodrigue, 2001; Sencindiver and Ammons, 2000; Zeleznik and Skousen, 1996).

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 a thermal analysis technique. The linear regression equation (R2=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:

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

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 hectare (Mg C ha^{-1}) using the bulk density, coarse fragment content, and layer depth data for each soil pit in the data set from Rodrigue (2001).

Table 2-1. 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 hardwoods	Tree Basal Area pines ‡	Total Tree Basal Area
		years	meters		---- % of total ----		cm^2
				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 2-1 (continued).

State	Site	Stand Age †	Site Index	Canopy Type	Tree Basal Area hardwoods	pines ‡	Total Tree Basal Area
		years	meters		---- % 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	WV 1 - F3	31	17.4	E. white pine	31	69	10,681
(north)	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*
				NON-MINED LAND (adjacent to mined sites)			
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 2-1 (continued).

State	Site	Stand Age †	Site Index	Canopy Type	Tree Basal Area hardwoods	pines ‡	Total Tree Basal Area
		years	meters		---- % 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
	Average	*71*	*27.3*		*100*	*0*	*14,194*
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
	Average	*62*	*25.5*		*100*	*0*	*13,106*
(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
	Average	*51*	*25.9*		*100*	*0*	*12,291*

† 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).

Geostatistical Analysis

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

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

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 C Sequestration on Mined and Non-Mined Forest Land

For all measurement plots, we estimated above-ground tree biomass and coarse-root biomass (kg dry weight) 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 C 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 C 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 2-1), we defined each plot as follows: pine stands were defined as forest stands in which the basal area of all hardwood trees (BAHW) was lower than or equal to 20% of total stand basal area, i.e. BAHW < = 20%; mixed stands were defined as forest stands with BAHW ranging from 20 to 80%; and hardwood stands were defined as forest stands with BAHW > = 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 C sequestration potential of the mined sites. Standard statistical tools, the PROC CORR procedure, were used in SAS® to perform these analyses.

We performed multivariate regression analyses of ecosystem C content by forest type measured on mined land to determine the effects of site quality and stand age on the C sequestration potential of mined land under various forest cover. We used standard regression procedures (PROC REG and the Mallow's 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 multivariate 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(CEcosystem) = natural log of ecosystem carbon content (Mg C ha^{-1}); b_i = regression coefficient, i = 0,1,2,…,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 (3-D) 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 3-D 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 C 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(Core10) = 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), *P. taeda* L. (loblolly pine), *P. sylvestris* L. (Scots pine), *Robinia pseudoacacia* L. (black locust), *Prunus serotina* Ehrh. (black cherry), *Quercus alba* L. (white oak), *Q. 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), *A. 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 (*Q. 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 reported in Rodrigue (2001, his Appendix E), 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(Core10) = -0.2856 + 0.9579 (1/DBH10) (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 on 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 (12 in) stump height to 10.16 cm (4 in) 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 C content from the second rotation, including stem wood, stem bark, foliage, treetops, branches, stumps, and coarse roots, to the C amount stored in wood products in use and in landfills manufactured from harvested tree biomass from the first rotation (estimated as 24% of the C 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 C pools were not in equilibrium and that C was being sequestered in the soil and litter layer biomass at rates similar to afforested lands. We assumed that C 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 C accumulation rate (Mg C ha^{-1} $year^{-1}$) could be approximated by dividing the present soil C content (Mg C ha^{-1}) by the stand age (years) in each mined plot. We assumed that the equilibrium state of soil C on forested mined sites could be approximated by the soil C levels measured on the adjacent, non-mined forest stands. For pine and mixed forest stands, we assumed that C would continue to accumulate in the soil C 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 C 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 C accumulation rate, Mg C ha^{-1}, by 60, or by assigning the equilibrium soil C content measured on adjacent, non-mined land, whichever was smaller.

For all mined plots, we assumed that the amount of C 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:

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

where A = coefficient representing the upper limit of litter layer C 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 C (Mg ha^{-1}) and stand age are derived experimentally. For example, for a given ecosystem, if the upper limit of litter layer C was represented by coefficient A', and the ecosystem-specific data point was (L', age'), where L' is the litter layer C mass (Mg ha^{-1}) measured at stand age age', then B could be computed as (A*age'/L') – age'.

We adopted the litter layer C mass decay equation developed by Smith and Heath (2002) to predict the amount of residual litter layer C in pine and mixed stands following harvesting disturbance at age 30 years. We estimated the amount of litter layer C 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):

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

where C = litter layer C 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 C 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 C 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 C 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:

$$A_i = \underset{i=1,\, j=1}{\overset{3,N}{Maximum}}\left(\frac{LitterC_{ij}}{Age_{ij}} * Mature\_years_i\right) \qquad \text{Eq.3}$$

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

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 C 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 C 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 C 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 C Sequestration at Rotation Age for Coniferous and Deciduous Forests on Mined Land

For a 60-year-planned C 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 C data sets for total tree, litter layer, and soil C 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, Mallow's 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) and 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 C 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 C 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 C 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 as a separate data point to use in standard statistical analyses, as opposed to using a limited number, stand-level, average C 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) (Fig. 2-2). To describe the auto-correlation of spatial data, researchers often model the ?(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 (Fig. 2-2) 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 ?(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 (Fig. 2-2). 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 2-2). The graphical representations of the results from Table 2-2 are presented in Appendix A.

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 (SI and Litter C) in hardwood stands, and all but two 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 2-2). 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 2-2).

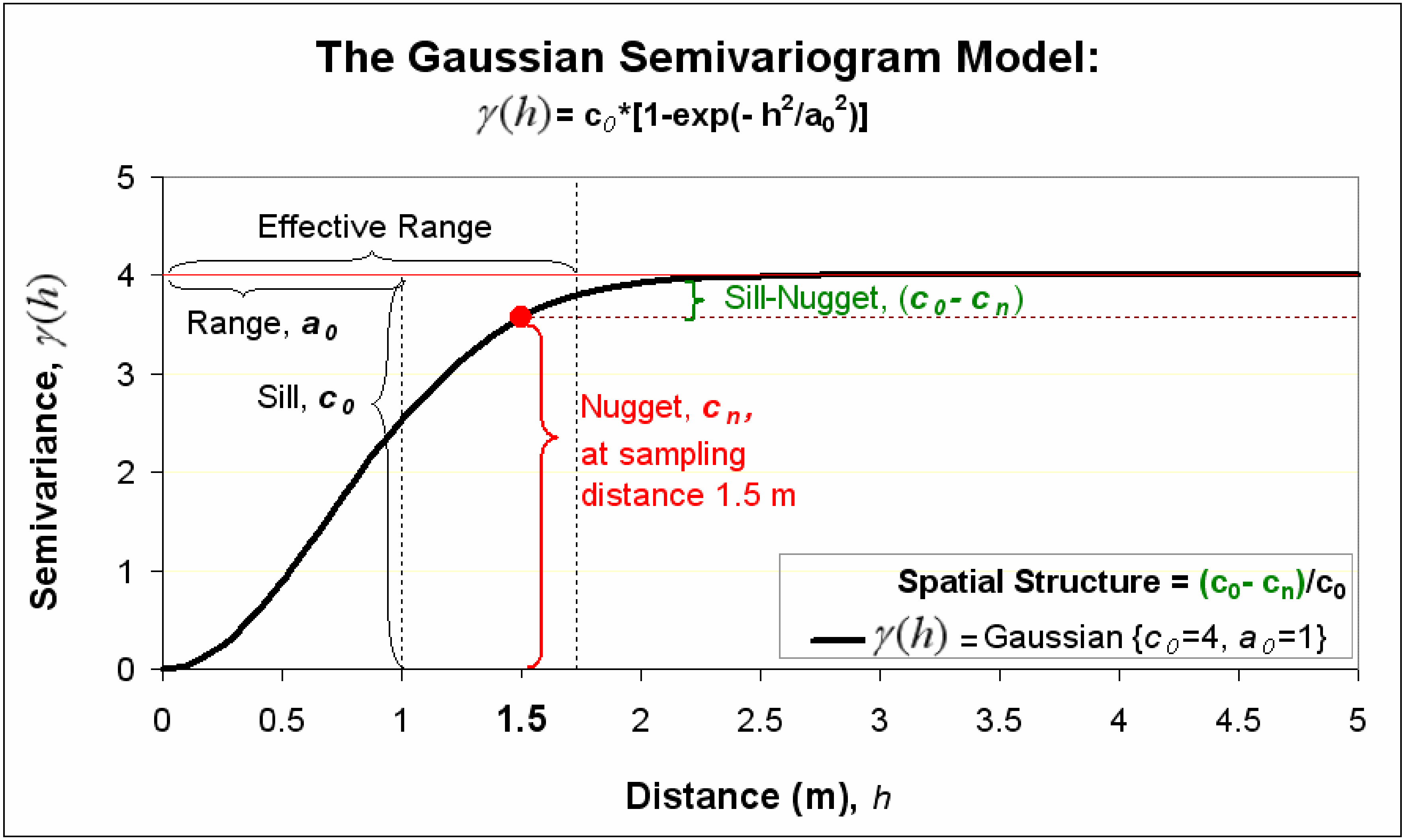

Figure 2-2. Components of the gaussian theoretical semivariogram model used to describe the presence or absence of auto-correlation of certain spatial data. The theoretical model with best visual fit to the semivariance statistic of the data was used for spatial data analyses and interpolation, such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

Table 2-2. 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. Graphical representations of the semivariances of these data are presented in Appendix A.

Forest Stand Type	Ecosystem property; C pool [a]	Gaussian model [b] c_0	Gaussian model [b] a_0	Spatial structure [c] at median distance	Distance sampled (m)
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)	--	--	--	

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 95,000 to 642,000 cm^2 ha^{-1} along a site index gradient that ranged from 15 to 34 m at stand age 50 yr (Fig. 2-3). 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 (Fig. 2-3) 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.

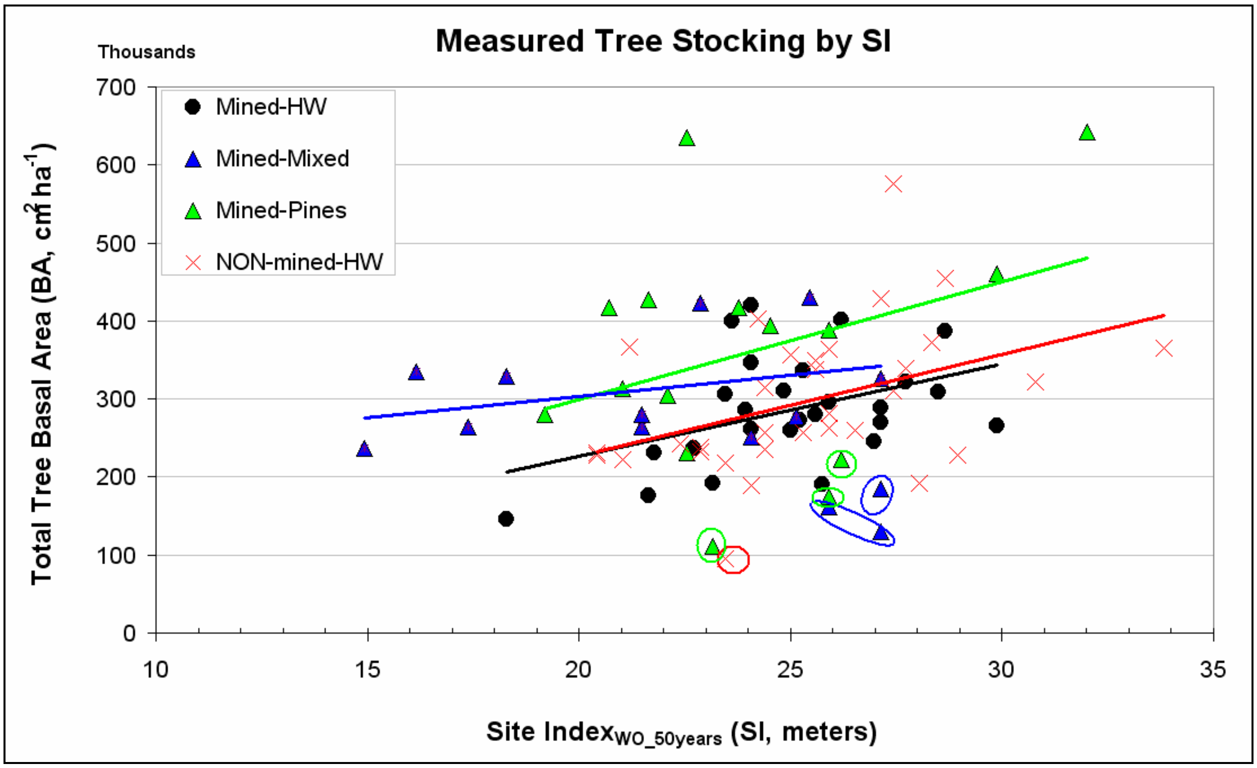

Figure 2-3. Basal area (tree stocking) of all measured plots on mined hardwood (Mined-HW), mined mixed, mined pine, and non-mined hardwood (NON-mined-HW) forest stands in the Midwestern and Appalachian coalfields. Regression lines indicate expected trends of stocking with increasing SI. Site index is the height, in meters, for white oak at base age 50 years. Circles indicate the plots identified as data outliers.

In order to develop a first approximation model of C 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.

The data outliers were IN 1-D1, IN 2-D3, and IN 2-F10 plots in Indiana from the mined mixed stand data set; VA-P1, -P2, -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 2-1, Fig. 2-3). 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 (Fig. 2-3). All of 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, -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 2-1, Fig. 2-3) 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 2-4. 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 is 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 C 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 2-3). 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.

Table 2-3. 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.

Forest Type	State / Site	Total Tree Carbon	Soil Carbon SOC	Soil Carbon WB-C*	Litter Layer Carbon	Ecosystem Carbon	**Projected Ecosystem C at 60 years**
		---- Mg ha^{-1} ----					
		MINED LAND					
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***	***168.4***
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 2-3 (continued).

Forest Type	State / Site	Total Tree Carbon	Soil Carbon SOC	Soil Carbon WB-C*	Litter Layer Carbon	Ecosystem Carbon	**Projected Ecosystem C at 60 years**
		----- 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***

NON-MINED LAND (adjacent to mined sites**)**

Forest Type	State / Site	Total Tree Carbon	Soil Carbon	Litter Layer Carbon	Ecosystem Carbon	Projected Ecosystem C at 60 years
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 2-3 (continued).

Forest Type	State / Site	Total Tree Carbon	Soil Carbon SOC	Soil Carbon WB-C*	Litter Layer Carbon	Ecosystem Carbon	**Projected Ecosystem C at 60 years**
		---------- 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.

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 2-3). Carbon associated with tree biomass of hardwood forests in Indiana was measured by Kaczmarek et al. (1995). For all but four non-mined and mined hardwood plots in Indiana and western Kentucky (Table 2-3), 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 2-3). 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).

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 2-3). Most of these new forests contained valuable species and trees were well spaced (Rodrigue, 2001; Rodrigue et al., 2002). Our data show 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 under-stocked, 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 2-3). 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 2-3).

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 2-3). 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 the non-mined soil carbon pool. Non-mined-site soil carbon ranged from 42 to 137 Mg ha^{-1} (Table 2-3). 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 2-3).

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 2-3). However, in a later study, Akala and Lal (2001) 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 (2001) 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 2-3).

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.

The original uncorrected Walkley-Black, and the corrected soil organic carbon estimates for the mined sites in this study, are reported in Table 2-3. On average, the SOC over-estimation of Walkley-Black carbon values ranged from 138% (in Ohio) to 236% (in Illinois) (Table 2-3). The correction equation, described in the Methods section of this paper, was based on data from carefully constructed and homogenously 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 (2001), can be successfully corrected for coal content.

Of the three ecosystem components, the soil had the greatest potential for sequestering and storing additional carbon. Mined land researchers often discuss the concept of "an empty cup" related to the potential of forest mine soils to 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 2-3).

These results indicated that the soil C pools on the mined lands had not reached equilibrium state (Fig. 2-4). Assuming that soil C 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 C equilibrium levels of the adjacent, non-mined forests. These estimates agree with the findings of Akala and Lal (2001), who reported that reclaimed forest land in Illinois could attain soil C 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 2-3).

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. 2-4, Table 2-3). 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 2-3).

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. 2-4, Table 2-3).

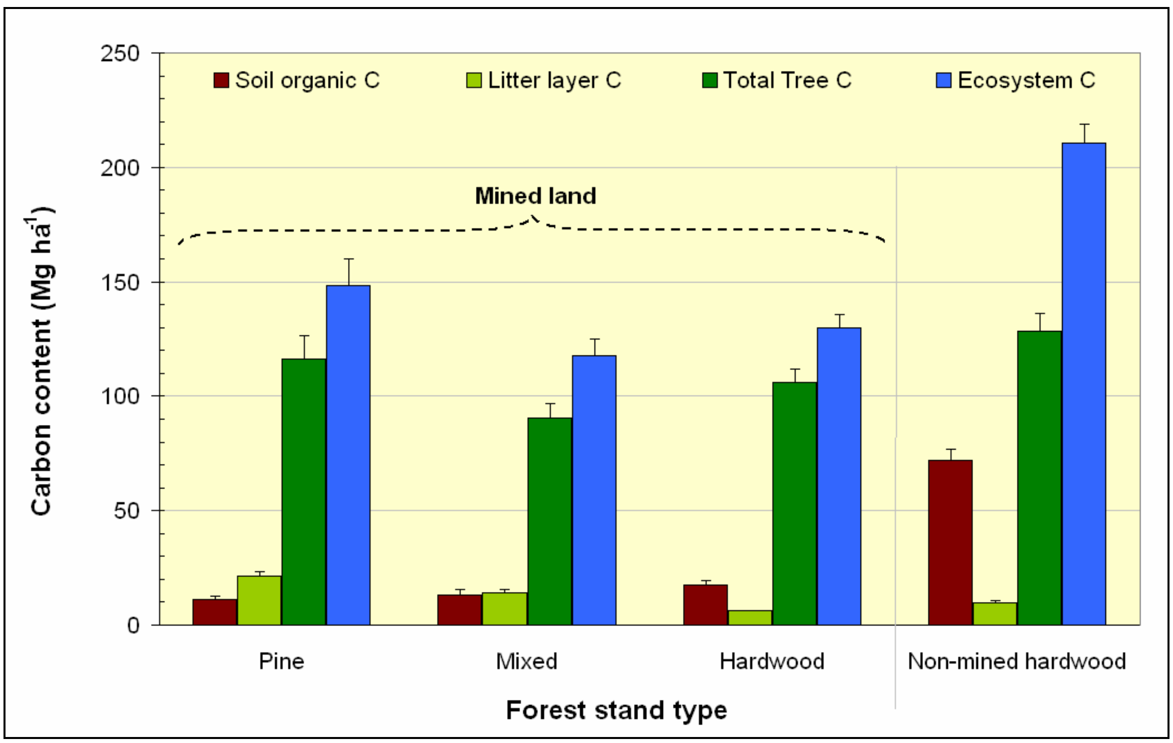

Figure 2-4. Carbon pools by ecosystem component, soil, litter layer, tree, and total ecosystem carbon content, for three forest stand types on mined land, pine, mixed, hardwood, and for non-mined hardwood stands, adjacent to the mined stands. Vertical lines represent one standard error of the mean estimate.

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, one-third of the total ecosystem carbon content (210 Mg ha^{-1}) was found in the soil (34%), 61% in tree biomass, and 5% in litter layer (Table 2-3).

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 2-3, Fig. 2-4). 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 2-3, the average rotation age ecosystem carbon content in non-mined stands was 205 Mg 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 2-3). In mined hardwood stands, the rotation age ecosystem carbon content ranged from 141 to 162 Mg 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 ha^{-1} in the decreasing order (by state) West Virginia (south) > Kentucky > West Virginia (north) > Indiana (Table 2-3).

Effects of Mine Site Quality and Stand Age on the C 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 2-4). 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 2-4. 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 coal fields. 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 SI	Age	SI/Age	n
			----- correlation coefficient & (P-value) ------			
Pine	Tree	*Ln*	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	*SQRT*	0.4250 *(0.1684)*	-0.3847 *(0.2169)*	0.4627 *(0.1299)*	12
	Ecosystem	*Ln*	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
	Litter	n/a	**-0.3606** *(0.0703)*	-0.1249 *(0.5431)*	-0.0435 *(0.8329)*	26
	Soil	*SQRT*	-0.2755 *(0.1731)*	-0.0935 *(0.6497)*	-0.0755 *(0.714)*	26
	Ecosystem	n/a	0.4944 *(0.0102)*	0.5675 *(0.0025)*	**-0.4294** *(0.0286)*	26
Mixed	Tree	*Ln*	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
	Soil	*SQRT*	0.8347 *(0.0014)*	-0.0477 *(0.8893)*	0.7299 *(0.0108)*	11
	Ecosystem	*Ln*	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 2-4). We expected the carbon content in all carbon pools to increase with the increase in site quality (i.e., higher SI value) because as trees capture atmospheric CO_2, 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 2-4). In contrast, the litter layer C 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 C in pine and hardwood stands (Table 2-4).

In pine stands, the tree carbon pool was significantly correlated with SI ($P = 0.0289$), Age ($P = 0.0820$), and 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 2-4). 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, $r = -0.5217$ and –0.4974, respectively (Table 2-4). 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 (Fig. 2-5). 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. 2-5 and 2-6). 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) (Fig. 2-6). 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 (Figs. 2-5 and 2-6).

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 C 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.

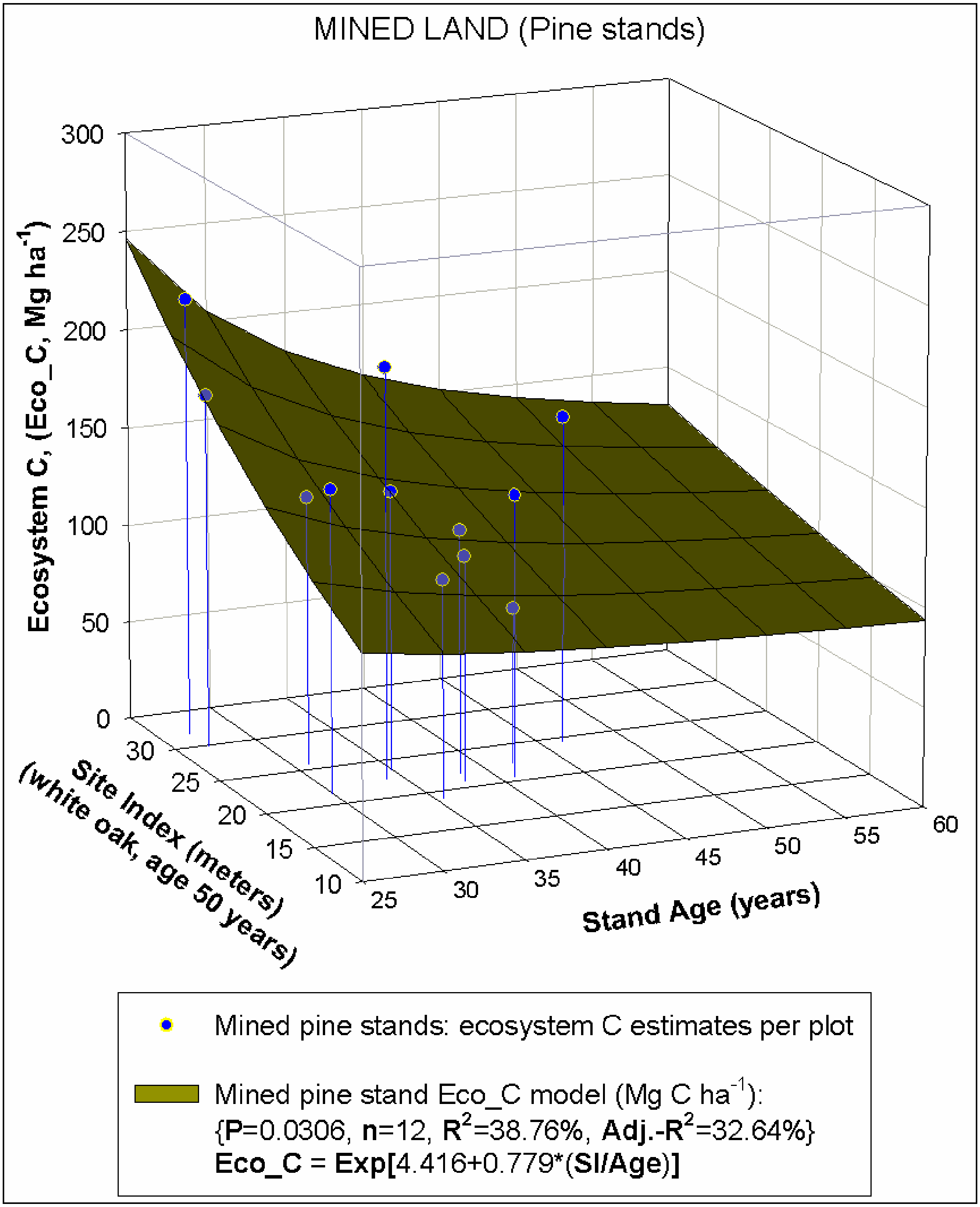

Figure 2-5. Ecosystem carbon content response surface by site index and stand age for pine stands on mined land in the Midwestern and Appalachian coalfields. Site index is the height for white oak at base age 50 years.

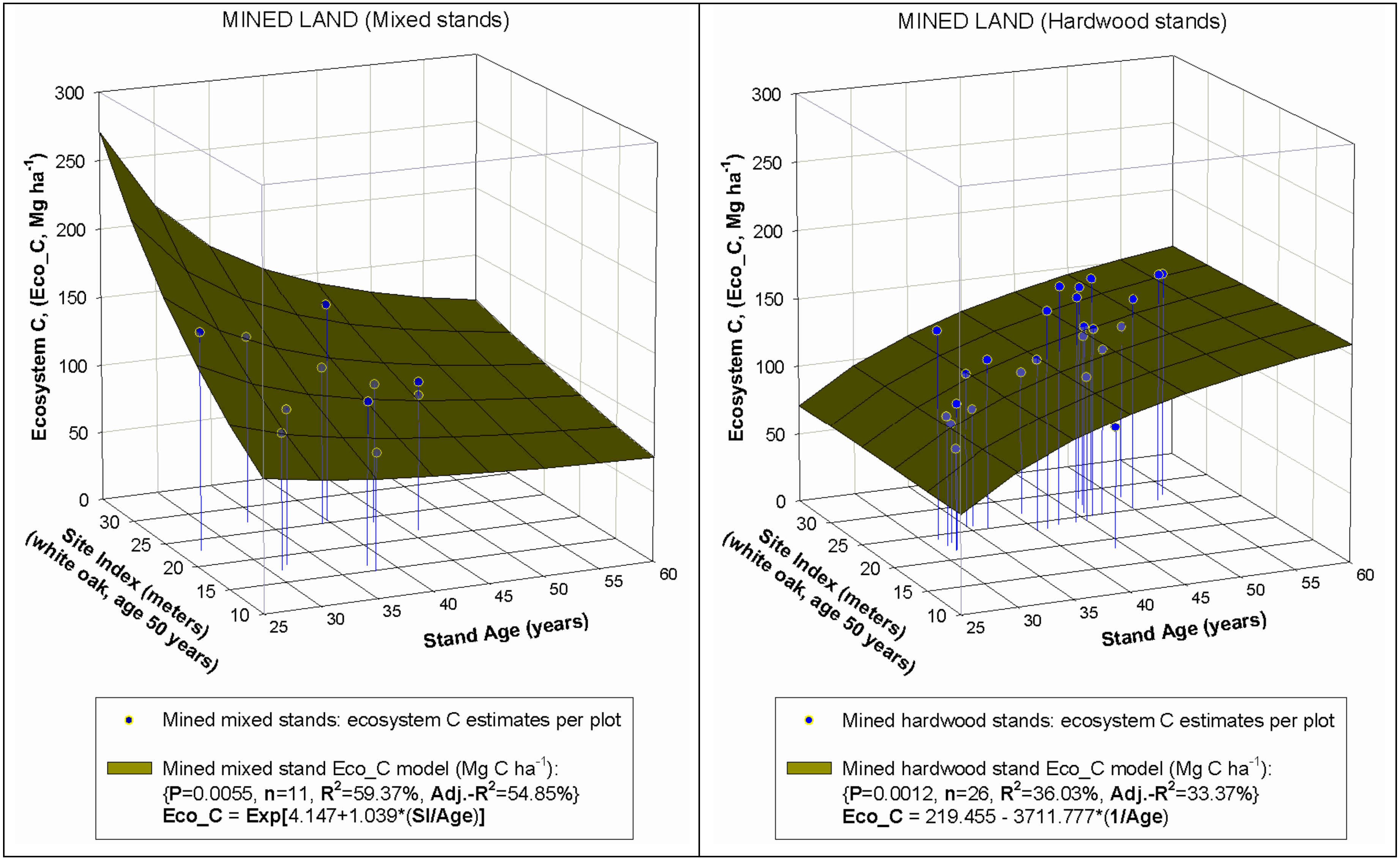

Figure 2-6. Ecosystem carbon content response surfaces by site index and stand age for mixed (left) and hardwood (right) forest stands on mined land in the Midwestern and Appalachian coalfields. Site index is the height for white oak at base age 50 years. Hardwood plot OH 2-B7 (age 71 yrs) was used in the analyses but is not depicted on the right-hand side 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 2-4). 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 (Fig. 2-6). 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 2-4), 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. 2-6). Evident from the regression equation depicted in Figure 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. 2-6). 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 counterintuitive 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 2-4). 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 2-4), the ratio of SI/age was positively correlated with the tree, soil, and ecosystem carbon pools (Table 2-4). The graphical representation of the ecosystem carbon response surface by SI and age is presented in Figure 2-6. 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. 2-5 and 2-6). 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. 2-5). Similarly, ecosystem carbon content for mixed stands increased exponentially with the increase of SI.

Empirical Models for Total Tree, Litter Layer, and Soil C Sequestration at Rotation Age 60 Years for Coniferous and Deciduous Forests on Mined Land. Comparison of Sequestered C 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. 2-7). We also compared mined and non-mined carbon content models at rotation age by ecosystem component in hardwood stands. (Fig. 2-8).

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. 2-7). 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, Eco_C'= ln(Eco_C), and the regression model was of the form Eco_C'= f(SI), then the regression model for Eco_C would be written as Eco_C = Exp[f(SI)], where Exp is the exponential function. We used the non-transformed C 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. 2-7).

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. 2-7).

Ecosystem C 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. 2-7), which is reflected in Figure 2-6. With an average estimate of 153 Mg C ha^{-1}, the 60-year estimate of Eco_C was significantly lower than the estimates for pine stands on high quality sites, SI > = 28 m (Fig. 2-7). 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 C associated with two 30-year rotations compared to one 60-year 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.

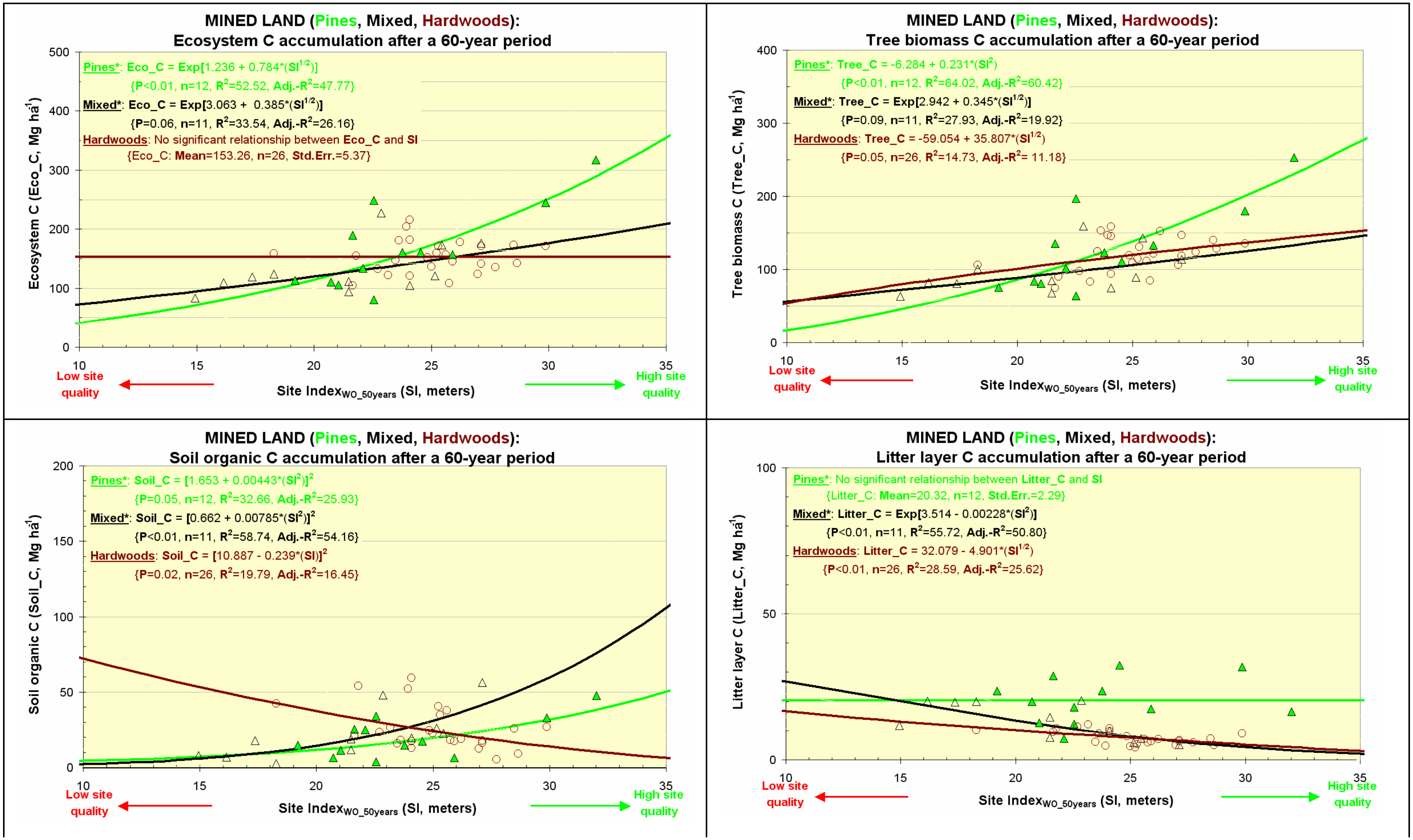

Figure 2-7. Regression models for total ecosystem, tree, soil, and litter layer carbon content projected to rotation age 60 years on forested mined land for three forest types, pine, mixed, hardwood, in the Midwestern and Appalachian coal fields. NOTE: Models followed by * were developed without data 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.

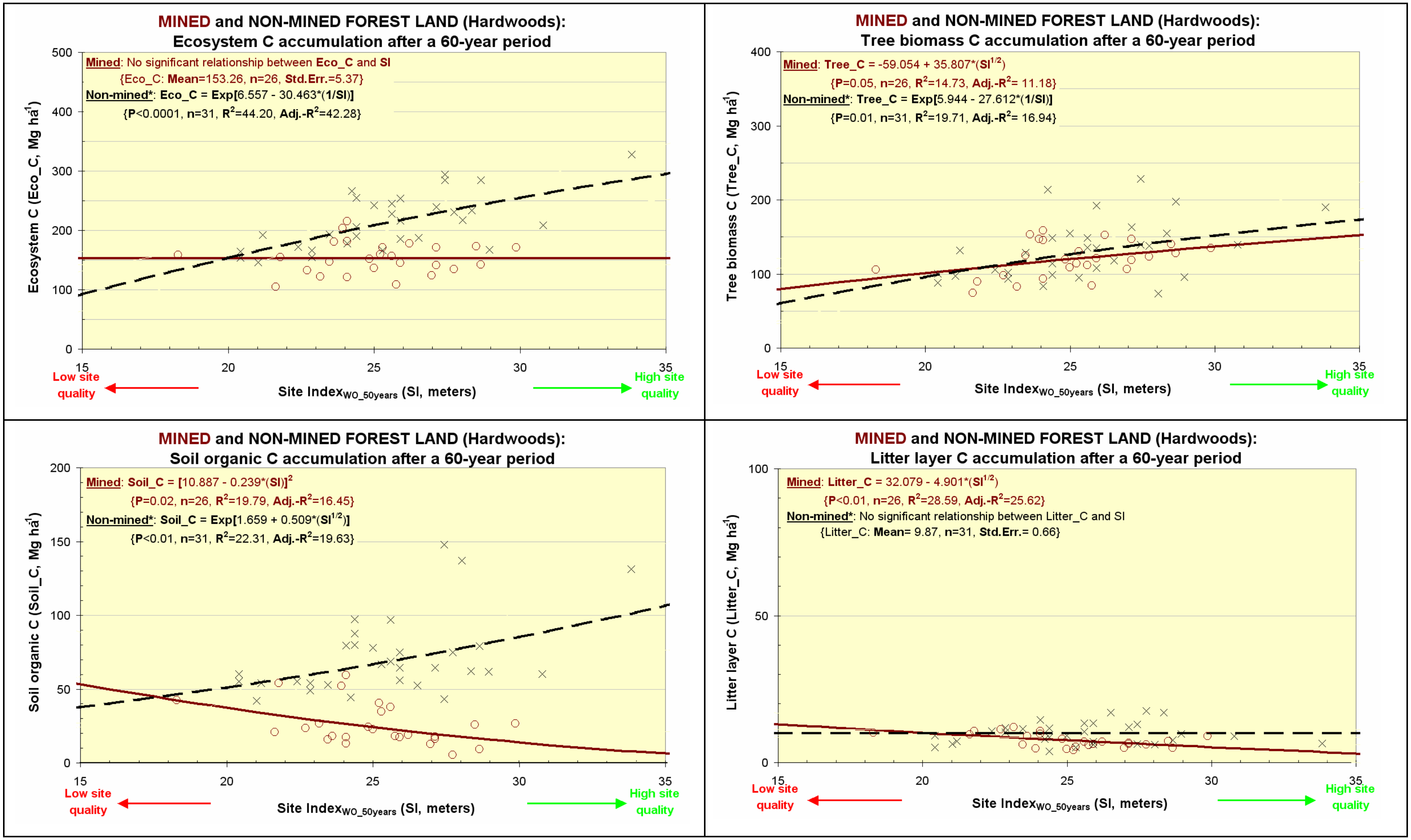

Figure 2-8. Comparison of regression models for total ecosystem, tree, soil, and litter layer carbon content projected to rotation age 60 years in hardwood stands on mined and non-mined land in the Midwestern and Appalachian coal fields. NOTE: Models followed by * were developed without data outliers, 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 appeared counterintuitive at first, the negligible cumulative effect of SI on Eco_C was reasonable considering the 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. 2-7). 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 CO_2 at higher rates. As a result, the cumulative effect of site quality on the ecosystem carbon content under hardwood vegetation was negligible (Fig. 2-7). 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. 2-7).

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 Figure 2-7, 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 2-9 we compared the ecosystem carbon model for mined mixed stands under two management scenarios, two 30-year rotations (2 x 30-yr) and one 60-year rotation (1 x 60-yr). The model showing ecosystem carbon content after two 30-year rotations was the same as depicted in Figure 2-7. The ecosystem carbon estimates for the 60-year rotation scenario for mined mixed stands were the same as depicted in Figure 2-6. The 3-D response surface in Figure 2-6 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 > 12 m), the Eco_C estimates for the 2 x 30-yr scenario were higher compared to the 1 x 60-yr 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 30-yr 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 60-yr 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 30-yr and 1 x 60-yr management scenarios, respectively (Fig. 2-9).

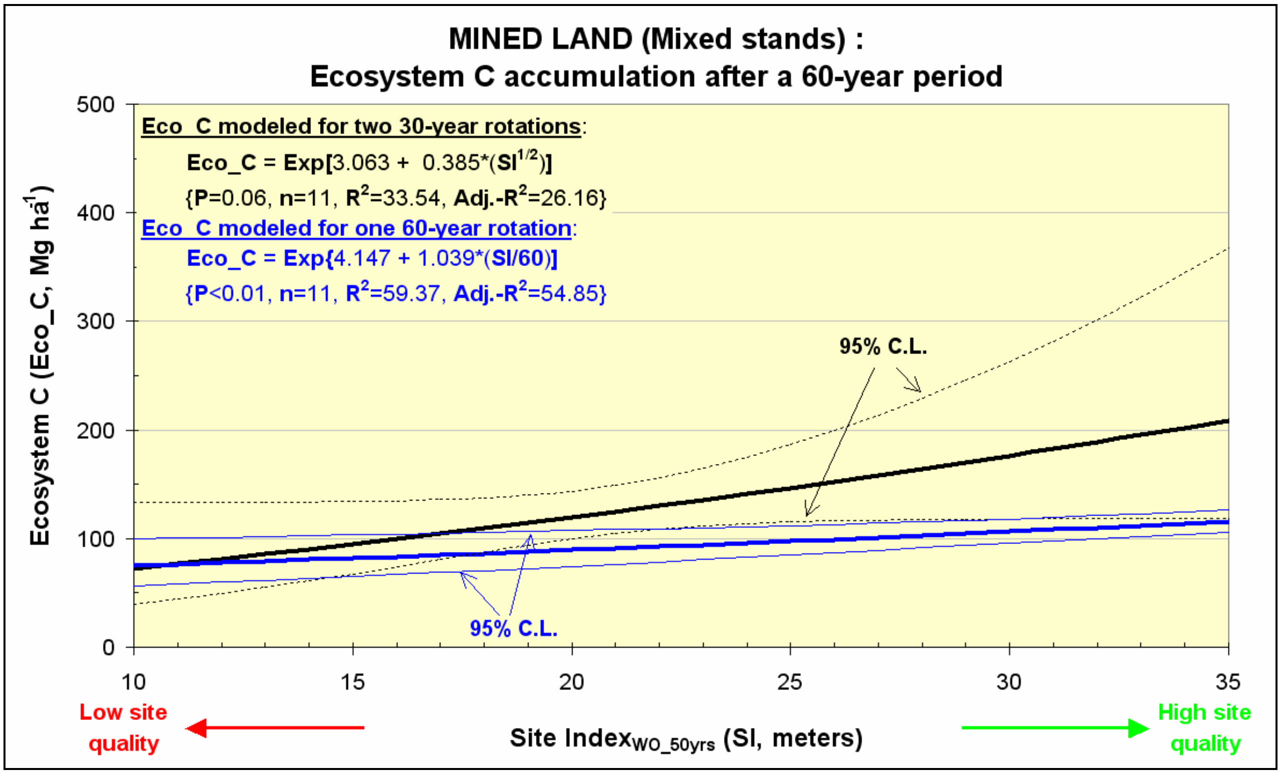

Figure 2-9. Comparison of regression models for total ecosystem carbon content projected to rotation age 60 years for two forest management scenarios, two 30-year rotations and one 60-year rotation, for mined mixed stands in the Midwestern and Appalachian coal fields. Site index is the height, in meters, 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 that the 30-year rotation for mixed stands would be the better management scenario when maximizing carbon sequestration is the main objective. These results should be used carefully, as they include no data analysis about the market value of sawtimber (for 1 x 60-yr scenario) and pulpwood (for 2 x 30-yr 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 above-mentioned 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. 2-7) supported our earlier findings from the correlation analysis presented in Table 2-4; 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. 2-7). 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. 2-7). 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 (Fig. 2-7). 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 (Fig. 2-7). 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. 2-7), 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. 2-7). 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. 2-7), it is clear that pine stands managed on 30-year rotations could sequester more carbon than mixed or hardwood stands if mined sites are restored to higher quality. On the other hand, 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. 2-8). 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 site 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. 2-8). 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 were 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. 2-8).

The results of the comparison of the carbon pools between mined and non-mined hardwood stands in Figure 2-8 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 C 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. If coal mining decreases site quality, it will decrease the C sequestration potential of previously forested mined land, despite the observed similarities of tree growth and litter accumulation rates before and after mining. 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 the long-term C sequestration potential is to be restored and maintained after mining.

Conclusions

We characterized 14 mined and eight adjacent, non-mined forests throughout the Midwestern and eastern coal regions to determine the C 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 C 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 C 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 Eco_C in both pine and mixed stands, and these models showed Eco_C decrease with stand age due to natural pine tree deterioration occurring in older pine stands and in the pine component of mixed stands. Ecosystem C 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.

Our results indicated that pre-SMCRA surface coal mining procedures used in the Midwestern and Appalachian coalfields of the U.S. could degrade forest site quality and could jeopardize the long-term productivity of forested mined lands; hence, could drastically reduce 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 mined hardwood stands. 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 C sequestration in total tree biomass, litter layer, and soils for pine, mixed, and hardwood forests established on mined land in the Midwestern and Appalachian coalfields. These equations could be used as building components of a decision support system (DSS) tool 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 such DSS tools 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 the society.

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Project Objective 2:
Effects of Silvicultural Treatments on Survival and Growth of Trees Planted on Reclaimed Mined Lands in the Appalachians

Introduction

All surface mining activities conducted after August 3, 1977, are subject to the provisions of Public Law 95-87, the Surface Mining Control and Reclamation Act of 1977 (SMCRA). This law mandated that reclamation be carried out on all surface mined lands, and set forth criteria for mine operators to follow in carrying out reclamation practices. Unfortunately, many of these criteria have created adverse conditions for reclamation with trees, and consequently reforestation of surface mined lands has decreased since the passage of SMCRA (Ashby 1991). These adverse conditions include: (1) excessive competing vegetation; (2) soil compaction; and (3) unfavorable soil chemical properties.

Competing vegetation is a direct result of ground covers sown to prevent soil erosion on newly reclaimed surfaces. The most commonly used ground covers include tall fescue, clover species, and other grasses and legumes. These dense grasses and legumes compete with tree seedlings for light, water and nutrients (Ashby 1991). On a surface mine in Indiana, Andersen and coworkers (1989) found that black walnut (*Juglans nigra*) and northern red oak (*Quercus rubra*) survival after seven growing seasons increased from 4% and 1%, respectively, when planted into an existing dense ground cover to 66% and 48%, respectively, when planted after ground cover was controlled. Adequate stocking of trees required to meet the specifications of SMCRA (1110 trees ha^{-1}) was only attained when the ground cover was controlled with herbicide. Height growth was significantly better on the mine site where weeds were controlled. Twelve-year results of this study again showed that height growth was enhanced by weed control (Chaney et al. 1995).

Soil compaction on post-SMCRA mined lands is also widespread. Soil compaction in mine soils is usually caused by the passage of large equipment over the soil in an effort to stabilize the soil when returning it to its approximate original contour as required by SMCRA. Soil compaction inhibits root growth of seedlings by increasing bulk density and consequently increasing soil strength, decreasing aeration porosity, and inhibiting the ability of the soil to drain once saturated (Omi 1985). Tillage treatments can ameliorate the detrimental effects of compaction. Ashby (1997) found that the mean height of 16 different tree species was significantly greater five years after ripping a mine soil to a depth of 1.2 m. Another study found that after 12 years, ripping to a depth of 85 cm significantly increased the survival, height, and diameter growth of both red oak and black walnut in southern Illinois (Ashby 1996). Black walnut seedlings growing on a surface mine in southern Illinois were found to have taproot lengths which were 92% and 75% greater in their first and second years of growth, respectively, in ripped versus unripped plots (Philo et al. 1982). This same study found overall rooting depth to be 81% and 58% greater in their first and second years in the ripped versus the unripped plots. For the second year only, radial root growth was found to be 89% greater in the ripped plots.

Chemical properties of mine soils are related to the overburden rock type from which these soils were created. In a study of the effect of overburden rock type on survival and growth of pitch x loblolly hybrids (*Pinus rigida* x *taeda*), an inverse relationship between soil pH and tree volume was found (Torbert et al. 1990). The rock types evaluated in this study consisted of pure sandstone and pure siltstone as well as mixtures of various amounts of these types. It is noteworthy that pH increased consistently as the proportion of sandstone decreased and as the proportion of siltstone increased. Plant available N and P are low on mine soils. Howard and coworkers (1988) found that mine soils in southwest Virginia had large quantities of P and K, but they suggest that P will likely be deficient even after fertilization due to the high P-fixing capacity of these soils. Another study in southwest Virginia found that compared to native forest soils, mine soils had less total N, and that the forms of N in the mine soils was largely unavailable to plants (Li and Daniels 1994).

Numerous species of trees have been studied for use in post-SMCRA reclamation and all with varying degrees of success, depending on the site conditions such as those previously described. White pine (*Pinus strobus* L.) has likely been the most extensively used species in reclamation. This species success in reclamation has ranged from good for a study in southwestern Virginia where the species averaged 58% survival and 3.8 m of height growth after 11 years (Torbert et al. 2000) to poor for a study in southeastern Ohio where no pines survived after three years (Larson et al. 1995).

Several hardwood species have also been tested for use in reclaiming post-SMCRA mined lands. Gorman and Skousen (2003) found excellent survival (90 to 100%) of several commercially valuable hardwoods on a reclaimed mountaintop removal mine in West Virginia when weed control and tillage were employed. A study in southwestern Virginia found 57%, 54%, and 91% survival for chestnut oak (*Quercus prinus* L.), yellow poplar (*Liriodendron tulipifera* L.), and white ash (*Fraxinus americana* L.), respectively, in plots where weeds were controlled chemically (Torbert et al. 1985).

The purpose of this study was 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. Additionally, selected physiological responses of hybrid poplar seedlings to these silvicultural treatments were assessed at one study site.

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). 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. 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). 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.

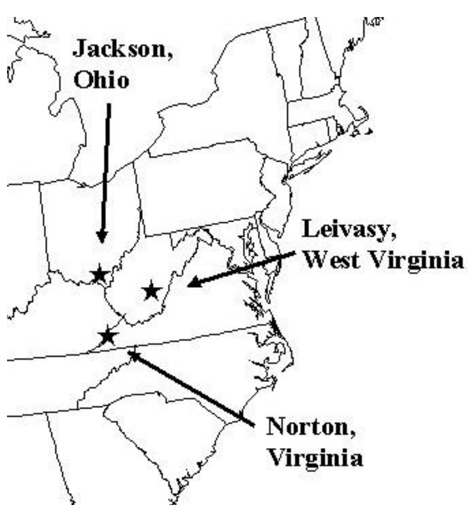

Figure 3-1. Location of study sites.

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 3-2 and 3-3). 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 3-2 and 3-3). 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 3-2 and 3-3). 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 3-2 and 3-3). 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 3-2. 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 3-3. 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 $4^{o}C$ 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 6*N* 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 3-2) 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 3-4). 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 3-5). 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 3-5). 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 3-5).

Table 3-4. 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 3-5. 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 3-4). 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 3-6). 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 3-6).

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 3-6). In WV, HP grew significantly more that 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 3-6).

Table 3-6. 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 3-4). 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 3-7). 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 3-7).

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 3-7). 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 3-7). 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 3-7).

Table 3-7. 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 3-4). 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 3-8). 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 3-8). 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 3-8. 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 3-4). 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 3-9). 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 3-9). 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 3-9), 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 3-9. 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 3-4). Analysis by site eliminated the species by treatment interaction in OH and showed no differences among treatments for volume growth at this site (Table 3-10). 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 cm^3) for WP compared to WC (2.3 cm^3), though neither treatment was significantly different from WC+T+F (2.8 cm^3). There was also a notable response for HP to WC+T (43.3 cm^3) and WC+T+F (51.7 cm^3) compared to WC (2.8 cm^3). 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 3-10).

Looking at species effects in OH, HP had significantly more volume growth than HW or WP across treatments (Table 3-10). 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 3-10).

Table 3-10. Average volume growth (cm^3) for research sites in Lawrence County, OH, Nicholas County, WV, and Wise County, VA, by treatment and species group.

Site and Treatment	Species Group HW	HP	WP	Treatment Mean
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 3-4). Analysis by site eliminated the species by treatment interaction for OH, where no treatment effects were evident (Table 3-11). 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 cm^{3}) than both WC+T and WC (54.1 and 15.6 cm^3, respectively) (Table 3-11).

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 3-11). 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 cm^3) followed by HW (25.9 cm^3) and WP (6.2 cm^3).

Table 3-11. Total tree volume (cm^3) for research sites in Lawrence County, OH, Nicholas County, WV, and Wise County, VA, by treatment and species group.

Site and Treatment	Species Group: HW	HP	WP	Treatment Mean
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 3-12). 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 3-13). 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 3-13).

Table 3-12. 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 3-13. 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 3-12). Analysis by site revealed that there were no treatment effects in OH across species groups, and all treatment means were negative (Table 3-14). 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 3-14). 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 3-14).

Table 3-14. 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 3-12). There were no treatment effects for any site or species (Table 3-15). 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 3-15).

Table 3-15. 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 3-2). 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).

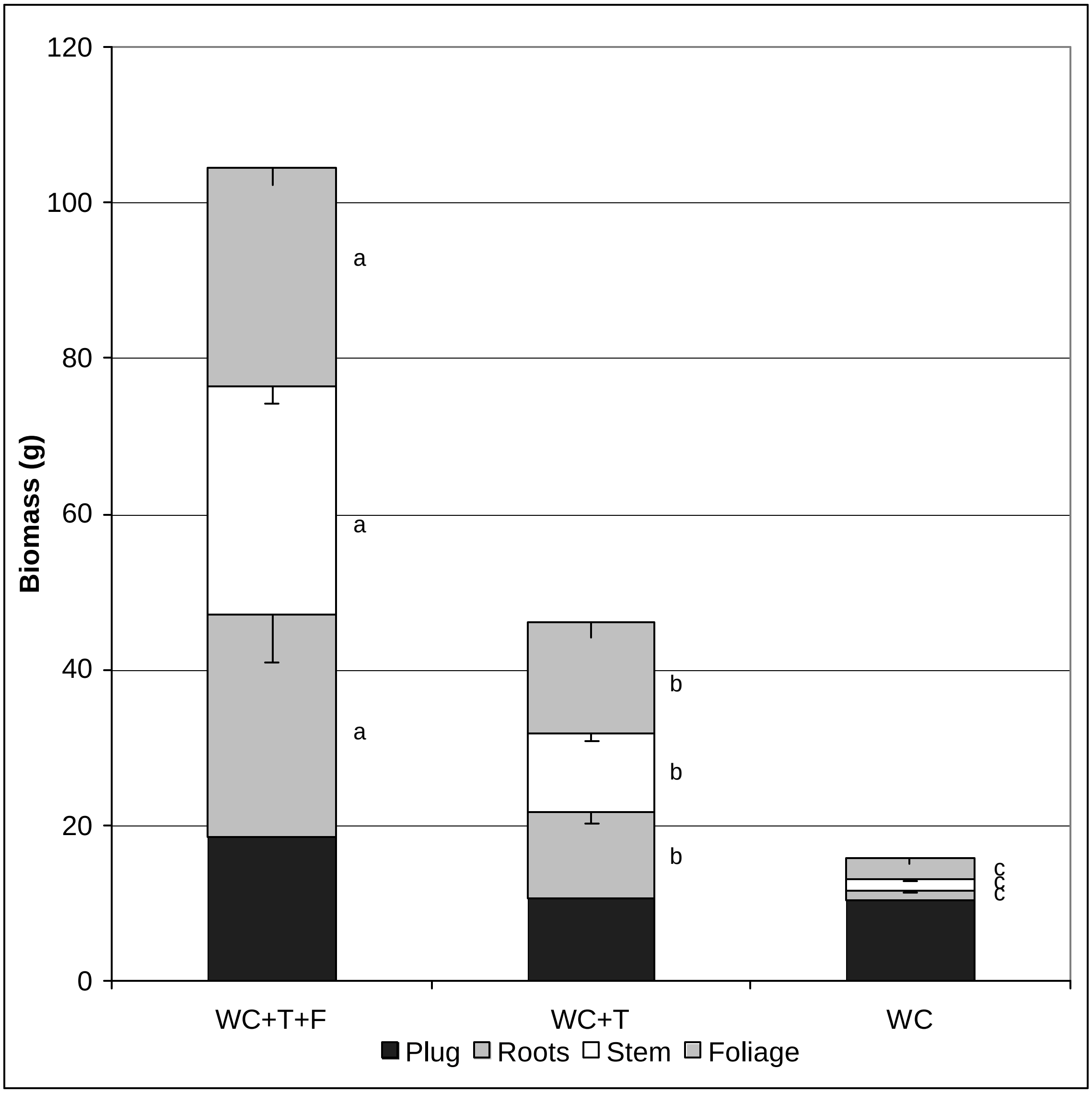

Figure 3-2. Hybrid poplar biomass by plant part and treatment for study site in Nicholas County, WV. Letters beside segments indicate significant differences at the P < 0.05 level among treatments for that particular segment.

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 3-16). 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 3-16). 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 3-16. 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 3-17).

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 3-17) 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 3-17. 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 3-4). 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 3-2 and 3-3). Both electrical conductivity and pH were within ranges (>0.05 dS m^{-1} and pH > 6.0, respectively) reported by Torbert and coworkers (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. 05dS 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 3-2 and 3-3). 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 cm^3 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

Successful reforestation of surface mined land requires selecting sites with suitable soil characteristics for good establishment and growth of trees. Soil conditions can be altered through silvicultural treatments to ameliorate conditions that limit tree establishment and trowth on these lands. Reclamation procedures that utilize sandstone overburden as the minesoil parent material, reduce compaction caused by the use of heavy equipment to grade the site, and establish tree compatible ground cover can reduce the need for these silivicultural treatments (Torbert and Burger, 2000).

Several specific 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.

The results of this investigation on reforestation husbandry practices show the importance of recognizing the interactions between site conditions, silvicultural treatments, and tree species to successfully establish trees on minesoils. The interactions among these factors determine the success or failure of reforestation efforts.

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