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Effects of pine-hardwood management practices on forest regeneration and woody species diversity at the Savannah River Site, South Carolina, USA

Kimberly K. Crider

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To the Graduate Council:

I am submitting herewith a thesis written by Kimberly K. Crider entitled "Effects of pine-hardwood management practices on forest regeneration and woody species diversity at the Savannah River Site, South Carolina, USA." I have examined the final electronic copy of this thesis for form and content and recommend that it be accepted in partial fulfillment of the requirements for the degree of Master of Science, with a major in Geography.

Kenneth Orvis, Major Professor

We have read this thesis and recommend its acceptance:

Accepted for the Council:
Dixie L. Thompson

Vice Provost and Dean of the Graduate School

(Original signatures are on file with official student records.)


To the Graduate Council:

I am submitting herewith a thesis written by Kimberly K. Crider entitled "Effects of Pine-Hardwood Management Practices on Forest Regeneration and Woody Species Diversity at the Savannah River Site, South Carolina, USA." I have examined the final paper copy of this thesis for form and content and recommend that it be accepted in partial fulfillment of the requirements for the degree of Master of Science, with a major in Geography.

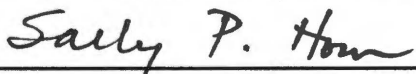


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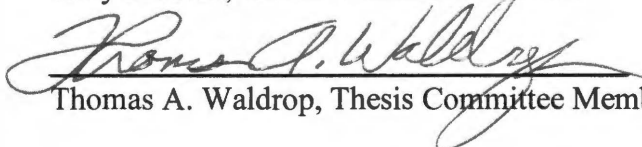
We have read this thesis and
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Acceptance for the Council:



Vice Provost and Dean of Graduate Studies

EFFECTS OF PINE-HARDWOOD MANAGEMENT PRACTICES ON FOREST
REGENERATION AND WOODY SPECIES DIVERSITY AT THE SAVANNAH
RIVER SITE, SOUTH CAROLINA, USA

A Thesis

Presented for the

Master of Science

Degree

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Kimberly K. Crider

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ABSTRACT

In 1989, mixed hardwood-pine forest sites at the Savannah River Site in South Carolina were chosen by USDA Forest Service employees for use in a study of the effects of a combination of forest management practices on woody species composition and diversity. The sites were surveyed for species composition, harvested commercially, burned using several severities, and planted with pine seedlings during 1990. In 1991 and 1993 the sites were surveyed again by Forest Service employees for post-disturbance species composition. I recovered and compiled the earlier pre- and post-disturbance data, and resurveyed the sites in 2002 to compare the immediate effects and the possible persistence of effects of the management treatments on woody species composition and diversity over an 11 year period.

The treatment combinations represented a range of disturbances: two harvesting treatments (dormant and growing season commercial clearcuts), three site preparation burn treatments (high severity burn, low severity burn, or no burn), and two pine regeneration treatments (planting of *Pinus taeda* L. seedlings and natural regeneration). Twelve sample plots, representing all possible treatment combinations, were established in each of three replications for a total of 36 plots. Woody stems were counted and identified in samples in each of the 36 plots at ages one, three, and 11 years. I compared density and diversity values (Shannon-Wiener diversity, richness, and equitability) among treatment combinations using analysis of variance (ANOVA).

The most significant effects on species density and diversity were evident one growing season after the harvest. Season of harvest had the most significant effect on both species densities and diversity, while burn severity had a significant effect on

diversity alone. At age three, the same patterns were detected, but were weaker and insignificant. By age 11, *Quercus* and *Carya* densities differed significantly between the two harvesting treatments, indicating that this treatment had a lasting effect on these taxa. Significant differences for the remaining taxa were no longer evident at age 11. The replicates themselves (physically similar at the time of establishment) were significantly different, in many cases, when considered independently. This indicates that physical site characteristics, likely related to pre-harvest species composition, might have had a stronger underlying influence on post-disturbance species composition and diversity than the management treatments.

Overall, the results suggest that mixed hardwood-pine forests in the Atlantic Coastal Plain (ACP) consist of species able to vigorously recolonize following disturbances as severe as clearcutting. Although these types of management disturbances might have immediate effects on woody species composition and diversity, the results suggest that these effects are minimal over time in the absence of additional disturbance. If the desired outcome is to minimize changes in diversity and species composition, these strategies can be employed with probable success in mixed hardwood-pine forests of the ACP. However, more research should be conducted that explores management disturbances that are more intense and frequent (e.g., conversion to pine plantations)—these types of practices are becoming more and more common in ACP ecosystems as the demand for wood and wood products increases.

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CHAPTER I.

INTRODUCTION

Purpose

Scientists generally agree that the current rate of loss of biodiversity constitutes a major problem in relation to the future of human welfare (OTA 1987; Wilson and Peter 1988). Certain disturbance events and anthropogenic activities such as agricultural use and land clearance for human habitation can negatively affect biodiversity with resulting unpredictable effects on local ecosystems that may eventually cascade into threats to Earth's life-support systems. To counter such threats, we must accept the challenge to sustain biodiversity by increasing scientific knowledge of the effects of human activity on species' distributions and numbers, and applying this knowledge to predict likely patterns of biodiversity under emerging scenarios of future environmental change (Lubchenco et al. 1991).

Biodiversity has become an increasingly focal issue among scientists, conservation planners, political decision makers, and the general public over the past century as environmental degradation becomes more obvious and alarming. In the late 1890s and early 1900s, public concern for conservation of forest resources engendered the formation of many environmental conservation organizations and programs that laid the basis for the transfer of forest reserves to the Department of Agriculture's Forest Service in 1905 (Sharitz et al. 1992). Programs and legislation have been developed since to promote biodiversity both directly and indirectly. These include the establishment of Research Natural Areas, the Multiple-Use Sustained-Yield Act of 1960, the Endangered Species Act of 1973, Vegetation Ecology, Tree Genetics Improvement, and New

Perspectives in Forestry (Salwasser 1989). The National Forest Management Act (NFMA) of 1979 requires the consideration of the diversity of plant and animal species and communities throughout the planning process. Additionally, the act requires that forest plans verify either the existing diversity or the planned diversity goal be consistent with the overall multiple-use objectives of the planning area. The result has been increased efforts to develop and critique alternative forest management schemes with the aim of preserving and improving biological diversity in managed forests.

Early clearance of southern forests for human habitation and agriculture and, later, intense forest management strategies to feed high demands for wood fiber and timber products have led to concerns about the long-term sustainability of southern forests. Fifty-five percent of timber removals in the United States occurred in the South as of 1993 (Powell et al. 1993). During the 1990s, the Atlantic Coastal Plain (ACP) was the primary source of wood fiber for the entire United States (Christensen 2000). Projections indicate that an increased demand for paper and paperboard will incite a facilitating increase in the supply of softwood pulp primarily from southern pine plantations up to and beyond 2010 (Ince and Durbak 2002). The increased demand will no doubt incite increased conversion of southeastern forests to pine monocultures that require intense management with detrimental effects on habitat quality and species diversity.

Alternatives to pine plantation management started being considered as a result of growing interest and concern over biological diversity and forest ecosystems in the mid-1980s. Management for the development of pine-hardwood mixtures was introduced at this time. The mixtures offer both economic and ecological advantages over pine

plantations (Cooper 1989). In this type of management, hardwoods are not controlled by herbicides or mechanical treatments, reducing the cost and negative effects of silvicultural operations on a site. Low-intensity site preparation techniques decrease the potential for erosion and protect long-term productivity (Lloyd and Waldrop 1997). In comparison with pine monoculture, species diversity is maintained because species are not selectively eliminated.

As harvesting pressure continues with no sign of abating, research is required to assess the effects of harvesting and other common management strategies on forest species diversity and ecology. Information gained from this research can guide practices that promote the future sustainability of both economically and ecologically valuable forest resources.

The Savannah River Site (SRS) in South Carolina has provided unique opportunities in biogeographical and ecological research. Of the 74,000 ha that the SRS occupies, 65,000 ha were set aside for management by the United States Department of Agriculture Forest Service (USDAFS). The benefits of management were described as (1) the use of "idle" land, (2) control of erosion and weed growth, (3) economic return to the government for the harvest and sale of pulpwood and sawtimber, and (4) improvements of existing forests (White and Gaines 2000). Thus, the SRS provides opportunities for assessing the effects of management strategies on various aspects of forest ecology.

The goals of this study are to analyze the effects of season of harvest, six variations of site regeneration burn treatments, and two pine regeneration treatments on

woody species diversity at the SRS after one, three, and eleven post-harvest growing seasons. The specific research questions were:

- 1) Do post-harvest species composition and diversity of woody species differ as a result of season of harvest?
- 2) Does the level of burn severity affect post-harvest species composition and diversity in burned plots?
- 3) Do stands that were planted with loblolly pine seedlings exhibit differences in woody species composition and diversity compared to those where pines were allowed to regenerate naturally?

The combination of site preparation treatments and time span of this project make it unique within the large body of existing literature relevant to this topic. A great deal of research has investigated the effects of site preparation burning on hardwood and pine species regeneration; however, very few studies attempted to clarify how different burn severities affect the composition and diversity of woody species. In addition, studies of the effect of season of harvest on hardwood regeneration in the interest of pine survival for the proliferation of pine monocultures have been numerous over the past few decades. Very few of these studies, however, have attempted to relate season of harvest directly to plant species composition and diversity. The combinations of management treatments that this study incorporates provide a unique opportunity to explore the changes in woody species along a longer temporal scale and a unique human-induced gradient of site disturbance.

Literature Review

Relevant Theoretical Work: Succession and Diversity

F. E. Clements was the first to propose a formal schema of plant community change, or succession (Barnes et al. 1998). Clements' work outlined distinct stages of forest development, from an initial bare site to a climax community. He described the process as a succession of groups of associated species that existed together and were eventually replaced by later-stage associations. Clements' theory, however, has been criticized for its orderly and deterministic view of succession wherein each vegetation step represents a stage toward the development of one entire "organic entity," a stable, self-reproducing climax community. Critics have argued that the theory does not consider the stability or instability of vegetation types and individual species that are not considered part of the predicted climax formations (Gleason 1917; Harper 1977; McIntosh 1981). The Clementsian view of succession has also been criticized for overlooking the fact that even the most stable association is never in complete equilibrium—impending disturbance or the influence of surrounding disturbed areas continually threaten the stability of the associations and the system as a whole.

The criticisms of Clements' holistic view of succession based on the "superorganism" climax eventually led to the formation of the individualistic or reductionist theory of succession introduced by Gleason (1926). This theory views plant communities as assemblages of individual species whose abundances are determined by properties of that species, such as longevity or the ability to partition resources (Lafon 1995). The Gleasonian approach considers the individual plant mechanisms that determine competition for resources, the consequent mortality of certain species, and

replacement by other species individually, rather than by entire assemblages or associations of species (Gleason 1926). Egler (1954) introduced a similar individualistic approach in the initial floristic composition model. According to this model, biological legacies in the form of propagules of most pioneer and late successional species exist on a site at the time secondary succession is initiated. The characteristics of these species and competitive dynamics between species determine which species will dominate a site. Short-lived species are eventually replaced by long-lived species. The reductionist ideas of Gleason and Egler have become dominant in the literature during the last few decades because they address factors relevant to secondary succession, primarily by taking into account the influence of biological and environmental legacies on the process of succession.

The development of ecosystem ecology renewed interest in the study of succession by recognizing the need to incorporate system-level functional properties of energetics, such as biomass changes, along with the more traditional structural aspects, such as species composition (Reiners 1992). Bormann and Likens (1979) developed a more recent view of succession often cited in current literature. This model is based on the functional properties of northern hardwood forests and divides forest development into four phases: (1) Reorganization, a period of 10–20 years during which the ecosystem loses total biomass despite the accumulation of living biomass; (2) Aggradation, a period of more than a century during which total biomass increases to reach a peak at the end of the phase; (3) Transition, a variable length of time during which total biomass begins to decline; and (4) Steady state, wherein total biomass fluctuates about a mean. Although this model was written with respect to northern hardwood forests of North America, it is

significant to succession theory because it provides a framework for relating life history traits of plants with environmental variables of a particular region to help predict responses to disturbance events (Reiners 1992). This model is also unique because it defines a temporal scale applicable to northern forests and provides a general timeline for summarizing the status of forest development based on age (i.e., time since disturbance). In addition, it is generally important to current field studies in succession because it emphasizes disturbance and its incorporation of local environmental variables and local species characteristics as major elements affecting succession.

Because changes in biodiversity are an integral part of forest succession, an ever-growing body of literature exists on species diversity changes during the course of succession (for reviews, see Billings 1938; Bazzaz 1975; Harcombe 1977). The characteristics of the unique events that initiate succession, the environmental conditions at the time of initiation, and the spatial and temporal scale being considered greatly affect diversity as a measured value as well as a concept. Therefore, it is important to define biodiversity within the context of the study that one is pursuing. Alpha diversity (within habitat diversity), beta diversity (between habitat diversity), and gamma diversity (evolutionary diversity or the differences in species composition between similar habitats in different geographical areas) are classes commonly used to distinguish between the scale-dependent levels of diversity. The choice of the definition used depends on the type of study and the spatial scale at which diversity is being quantified or described. Alpha and beta diversity are the most frequently used in studies of plant species diversity.

Current theories suggest that disturbance plays the main role in maintaining species diversity by preventing competitive dominance of one or a few species (Roberts

and Gilliam 1995). Disturbances can additionally increase the environmental heterogeneity of an area by providing diverse niches as a basis for specialization and resource partitioning by species (Denslow 1985). The effect of a disturbance on biodiversity depends not only on the properties of the disturbance itself, but also on the state of the ecosystem in terms of dynamics of population growth and competition both before and after the disturbance (Huston 1994). The amount of time that it takes for an ecosystem to recover is not only dependent on the interval between disturbance events (frequency), but is also greatly affected by the intensity of the disturbance (proportion of biomass killed) and the availability of critical resources in the environment. The inherent resiliency of the affected ecosystem also plays an important role in its recovery and is defined by the typical or historic disturbance regime of the ecosystem of interest. Along a gradient of increasing disturbance frequency or intensity, species diversity can increase, decrease, or peak in the middle of the gradient depending on the environmental conditions that influence the growth and recovery of a system from a disturbance (Huston 1994).

Relevant Field Studies: Succession and Diversity

The theoretical research presented above has been supported and refuted over the years by countless field studies that have examined succession in many different regions of the world and within various spatial and temporal contexts. The goals of these studies have most often been to understand the successional stages that typify vegetation growth on abandoned farmland, commonly referred to as old field succession (Lafon 1995). Unlike the current project, the aforementioned studies do not necessarily use land use or site history information to compare successional stages, but instead focus on age and

vegetational growth stages among similar areas. Although the specific goals of this thesis differ from those of old field studies, these studies are nonetheless fundamentally relevant to the current project in that the sample plots for the current project were essentially abandoned and allowed to develop naturally after clearcutting. Because species dominance and success are an integral part of succession, diversity is an innate component of these types of studies and is often presented in terms of species densities and basal areas rather than in the form of commonly used diversity indices (e.g., equitability and Shannon diversity).

Billings (1938) and others studied different stages of succession simultaneously by using space-for-time substitution in the North Carolina Piedmont. Billings found that shortleaf pine (*Pinus echinata* Mill.) was the first woody species to establish following herbaceous species invasion. He found that pine density increased for a short period until the pines began to mature, at which time density began to decrease. With the senescence of pines, hardwoods such as oak (*Quercus* spp.) and hickory (*Carya* spp.) became dominant (Billings 1938; Christensen and Peet 1981). Nicholson and Monk (1974) studied forests in the Georgia Piedmont, also using a space-for-time substitution that included forests ranging from zero to greater than 200 years in age. The authors found that diversity in each of the forest strata studied (canopy, understory woody stems, shrub woody stems, and ground layer woody stems) and major growth forms (herbs, woody vines, shrubs, and trees) increased rapidly during an establishment period and then at a decreasing rate through the remainder of succession. The authors calculated measures of diversity (richness and equitability) and found that both diversity components generally

increased at a high rate and then began to slowly decline later in succession in each of the four forest strata.

Similar succession studies have also been conducted in the ACP. This region is often dominated by *Pinus palustris* Mill. (longleaf pine), *Pinus elliottii* Engelm. (slash pine), and *Pinus taeda* L. (loblolly pine) monocultures, as a result of natural disturbance regimes and forest management practices. After abandonment of much of what had eventually become unproductive agricultural land in this region, pines were the first woody dominants to establish (Quarterman and Keever 1962). In addition, logging and the frequent natural and human-induced burning of forests provided for continued pine dominance on the ACP. However, with the increase of fire suppression activities to “protect” forests, many of the pine-dominated forests of the region became dominated by economically less valuable hardwood species. As a result, successional studies in the Southeast Coastal Plain region and elsewhere shifted from old field investigations to field studies that sought to discover the effects of various silvicultural site disturbances such as site preparation burning and specific harvesting methods on the structural and compositional development of forests.

Common Silvicultural Practices and Species Composition

Silvicultural terminology and definitions differ among regions and forest cover types. The science of silviculture is based on the control of forest composition, establishment and growth (Spurr 1945). Oliver and Larson (1990) have contributed much to the literature concerned with stand dynamics (the changes in forest stand structure with time), which includes stand behavior after disturbances.

Clearcutting has been used frequently to harvest mixed pine-hardwood (where pines dominate) and hardwood-pine (where hardwoods dominate) forests of the Southeast. Clearcutting of mixed species forests produces single aged stratified mixtures over time when advance regeneration, sprouts, new seedlings, or combinations of the three begin together following the disturbance (Smith et al. 1997). In these stands, the development of horizontal strata is created via the differentiation of heights of the different species making up the strata. Species groups of each stratum differ from the other groups in rate of height growth, tolerance of shade, rooting depth, and similar ecological characteristics. The dynamics of the succession of such stands can become very complex due to increased environmental heterogeneity created by the harvest itself in addition to heterogeneity created by any site preparation treatments that may have been applied after the harvest.

Site preparation burning is a commonly used technique following a clearcut. In the Southeast particularly, burning is used to prepare a seedbed for the regeneration of light-seeded species, such as pines, that rely on bare mineral soil for germination. At the same time, burning often eliminates sprouts of unwanted species and also controls heavy-seeded species that produce fewer seeds, depend on the burial of seeds beneath mineral soil, and have slower rates of germination than light-seeded species, by interfering with these processes (Smith et al. 1997). There are two components of fire: intensity, or the amount of energy released during the actual fire, and severity, the amount of biomass killed or consumed during the fire. In the mixed pine-hardwood forest of the coastal plain, low-severity burning was once a frequent natural disturbance (Ware et al. 1993). The frequent fires allowed for the growth of widely spaced pines with an open, grassy

understory and often very few mid-story or understory hardwood species. Thus, the practice of burning is geared toward both growing economically valuable pines, and reinstating the historic disturbance regime to achieve the goal of growing economically valuable pine forests. The use of frequent low-severity fires also provides habitat and reestablishes species associations that once existed by promoting the attributes of the historic habitat.

Site preparation burns can be applied at differing levels of severity, depending on the desired effect. The effects of the burn on particular species depend on that species' characteristics. Several authors have investigated the evolutionary adaptations of plant species to fire (e.g., Flint 1930; Mount 1964; Kauffman and Uhl 1990). Species with deep roots, thick bark, and buried buds are more resistant to fire than species with opposing attributes. In addition, many species in the Southeast, while they may not be completely fire resistant, often have traits that enable them to recover easily from fire. Barnes et al. (1998) summarize these traits, which include sprouting from the root collar (e.g., *Quercus* L. (oaks) and *Prunus serotina* Ehrh. (black cherry)), lignotubers, bole, or directly from the roots. Many coniferous species depend on fire for regeneration. Such species often have serotinous cones that open and release their seeds when heated to a certain temperature.

Because management procedures are typically very different from the disturbance processes that occur in natural forest stands, they have variable effects on diversity (Barnes et al. 1998). Clearcutting is arguably the most severe of forest management treatments. Stands that endure managerial treatment are often highly simplified and lack many components of community structure, such as snags and logs, as well as stand-level

structural complexity such as multiple canopy layers, gaps, and irregular tree spacing (Franklin 1995). Clearcutting not only reduces species diversity, but also has differential effects on different species. In the pine forests of the Southeast, this differentiation is especially apparent. In the absence of frequent fire, hardwood species come to dominate a clearcut site, whereas under a natural regime of frequent low-severity fire, pines would continue to prevail. As a result, added management in the form of frequent fires, thinning, and herbicide application must be continued to ensure the economic success of pines.

Relevant Field Studies: Forest Management, Disturbance, and Diversity

Many field studies were conducted in the early to mid-1900s that explored the relationship between herbaceous and woody species composition and forest management treatments. These studies were aimed at increasing pine production and understanding pine-hardwood successional dynamics in southern forests (Greene 1935; Wahlenberg 1935; Oosting 1944; Lemon 1946). During the early 1900s when naval stores operations were commonplace in ACP pine forests, fire was the main management tool used to encourage the growth of longleaf pine stands and discourage the survival of hardwoods and other less-valuable species. Even prior to this time, low-severity fire created by frequent lightning as well as by Native Americans is thought to have been the most frequent and therefore dominant disturbance agent in Coastal Plain forests, resulting in stands with little understory and unevenly spaced trees (Frost 1993).

As early as the late 19th century, Long stated that "... but for the continual annual wood firing that prevails so generally throughout the South, the Maritime Pine Belt would soon disappear and give place to a jungle of hardwood and deciduous trees" (1888, quoted in Heyward 1939). Heyward found that by the 1930s a region-wide interest in

forestry had resulted in complete fire protection of millions of hectares of land that had previously experienced a very short fire-return interval. Heyward found that hardwoods were numerous in all longleaf pine stands where fire had been excluded for more than 10 years. He also noted that as pines increased in height, with continued exclusion of fire, hardwoods were increasingly abundant due to the increased space beneath the pines that accommodated the canopy growth of shade-tolerant hardwoods. Furthermore, he recognized that although hardwoods were less economically valuable than pines, retention of some hardwoods in certain localities would benefit wildlife by providing a source of food and shelter.

Similarly, Hodgkins (1958) explored the link between fire and the composition of undergrowth in upland southern pine forests. While this study focused on effective hardwood removal techniques, it also provided insight into the mechanisms that facilitate oak regeneration following differing techniques of fire treatment. Hodgkins found that hardwoods were damaged more with hotter fires conducted in the late spring or early summer. He also documented that, in addition to season and severity of burning, physical site characteristics such as topography, and vegetation type at the time of burning, were factors that may affect future species composition. These findings suggested a general increase in diversity and growth of herbs and lesser woody species after prescribed fire in upland southern pine forests. Most importantly, the author noted that the succession pattern of forbs to perennial grasses to perennial woody species after fire seemed to depend more on the sprouting and seeding characteristics of the individual species involved than on the general succession of associations of species that the old field succession models suggested.

In a more comprehensive study, Wenger (1956) found that hardwood sprout growth rates after clearcutting were related to whether or not the site was burned, the season of harvest, and site-specific soil properties. Wenger found that robust hardwood sprouting had a detrimental effect on the growth of pine seedlings and saplings, and he related this robustness to future economic losses in pine production. Most important to my study was Wenger's suggestion that the aggregate response of all hardwood species was not likely to match the response of any one species. Thus, using such an aggregate measure would not accurately represent true species reactions to management treatments.

Literary views and general public opinion of clearcutting and other intense management practices such as burning, herbicide application, and soil scarification changed with changing public and scientific concern for preserving natural habitats and biodiversity. Subsequently, in the late 1980s and early 1990s, field studies that focused on the effects of timber harvesting and subsequent regeneration techniques on existing flora became more prevalent. Ecologists, environmentalists, and wildlife biologists expressed increasing concern that intensive techniques that discriminated in favor of a few preferred species might cause the local extinction of some plant species (Swindel et al. 1986). It was at this time that pine-hardwood mixtures were considered as an alternative to pine plantation management.

Felix et al. (1983) quantified the vegetation composition and structure of monoculture pine stands ranging from 1–22 years of age that had been converted from second-growth pine-hardwood forests in Virginia. The authors found that species richness and vegetation cover in the herbaceous stratum had high values in young stands of 1–5 years and declining values in stands from 5–15 years or at the point of canopy

closure. The authors concluded that although seral stages of old field succession were discernable, trends in species richness and evenness differed from old field succession models because a higher diversity of seeds and sprouts were present at the initiation of succession.

Swindel et al. (1986) conducted a study of regeneration patterns that followed two types of post-harvest site preparation treatments in slash pine plantations in northern Florida: a minimum disturbance treatment that involved very little relocation of the forest floor, and a maximum disturbance treatment in which severe dislocation of logging slash, the forest floor, and soil was carried out. The authors found that plant species richness and species diversity generally increased after harvest with differences in magnitude and pattern of response attributable to differences in treatment.

Zedaker et al. (1987, 1989) documented the development of mixed upland hardwood-pine mixtures after various harvesting and silvicultural treatments including: (1) commercial clearcut; (2) commercial clearcut and planting of *Pinus taeda*; (3) commercial clearcut, herbicide treatment, and planting of *P. taeda*; and (4) commercial clearcut, herbicide treatment of hardwood stumps, planting of *P. taeda*, and added release of pine seedlings via herbicide application to surrounding hardwood stems. Resulting tree species compositions ranged from: pure hardwood in the clearcut stands; hardwood-pine in the clearcut and planted stands; pine-hardwood in the clearcut, planted, and herbicide treated stands; and pure pine stands in the clearcut, planted stands with the double herbicide application.

McMinn and Nutter (1988) and McMinn (1992) studied the development of a low-quality oak-pine stand following harvest in two seasons and at two different

intensities. After 10 years, diversity and evenness (the proportions or relative abundances of a species over a given area) were higher after the growing-season harvest than after the dormant-season harvests. Diversity and evenness were also higher in the lower intensity harvest areas that were harvested in the dormant season. The authors relate these results to differences in seedfall characteristics between species and differences in soil disturbance created by the two harvesting intensities. They also found pine seedlings to be more abundant where harvesting was done with recently fallen seed on the forest floor, a larger proportion of the initial stand was removed, and more mineral soil was exposed by harvesting disturbance.

Greenberg et al. (1995) examined sites that had a greater variation in disturbance histories to determine if vegetation adapted to recovery from fire would respond similarly to other types of biomass removal. They compared *Pinus clausa* var. *clausa* (Chapm. ex Engelm.) Vasey ex Sarg. (sand pine) scrub stands with the following treatments: high-intensity burn, salvage logged, and naturally regenerated; clear-cut, roller-chopped, and broadcast-seeded; clearcut and bracke-seeded; and no treatment (mature forest). Their results suggested that overall richness and diversity of scrub vegetation were similar across different disturbance regimes, but that in some cases, single species responded to different treatments differently than cumulative results might predict.

Waldrop (1997) studied the effects of four pine-hardwood regeneration techniques on species composition in the Piedmont. Each of the stands was clearcut and then four different site preparation techniques were compared: spring felling of residuals over 1.5 m (5 ft) followed by summer broadcast burning, winter felling of residuals with summer broadcast burning, spring felling of residuals with no burning, and winter felling

of residuals with no burning. Measurements of number of seedlings and sprouts by species and heights of the dominant sprouts were taken at the end of each of the first four growing seasons and at the end of the sixth growing season. The results showed that postharvest species composition closely resembled that of the preharvest stands. The results also suggested that site preparation burning was not necessary to successfully establish pine-hardwood mixtures.

Jenkins and Parker (1998) compared resulting woody species composition and diversity in central hardwood forests of southern Indiana for two growing seasons after the application of four different silvicultural treatments: clearcuts, group selection cuts, single tree selection cuts, and 80–100 year uncut reference stands. Their results indicated that the smaller openings created by single tree selection cuts had much lower woody species richness and diversity than clearcuts, group selection cuts, and the reference stands. They concluded that a mixture of single and group selection cuts in addition to larger clearcut openings may be needed to maintain the woody species diversity of central hardwood forests by creating conditions that allow for the regeneration of both shade tolerant and intolerant woody species.

Fredericksen et al. (1999) studied the short-term effects of timber harvesting on understory plant communities on non-industrial private forestlands (NIPF) in two forest types (northern hardwood and oak-hickory) in Pennsylvania. They measured the species composition, richness, and diversity of herbaceous and shrub species on 40 NIPF stands harvested with different intensities (intensity was determined by the remaining basal area of each stand). The results indicated that short-term changes in herbaceous understory plant richness or diversity were affected by harvest intensity. However, slight changes in

vegetation structure (i.e., increased growth of forest-floor and shrub layers) and slight shifts in species composition were noted in both forest types with the most intense harvest treatments.

Scherer et al. (2000) measured the effects of post-harvest slash treatments on herbaceous species diversity. These treatments included a fall broadcast burn, a spring broadcast burn, a pile and burn treatment, a treatment that pulled all unmerchantable material using a cable system, a cleared treatment where slash was cleared and piled for the pile and burn treatment, and no treatment. The results suggested that the slash treatments in general increased the abundance of non-native herbaceous species and that the harvest itself reduced herbaceous species diversity. The slash treatments did not show any distinguishable effects on species diversity.

Gilliam (2002) analyzed the effects of timber harvesting on the herbaceous layer diversity of a central Appalachian hardwood forest in West Virginia. His findings showed that after 20 years of recovery following clearcutting, no significant differences in species diversity were discernible compared to mature stands. Gilliam discovered that, within the harvested stands, spatial relationships existed between herbaceous layer diversity and biotic (e.g., tree density) and abiotic factors (e.g., soil nutrients) and thus concluded that heterogeneity or microsite variations created by harvesting can have significant effects on the density and distribution of plant species within sites.

Brockway and Lewis (2003) studied the effects of deer and cattle grazing and timber harvest on plant species diversity and the long-term sustainability of longleaf pine ecosystems. They measured the diversity and cover of both woody and herbaceous species of stands that had been harvested in the early 20th century. The stands were either

thinned and not planted, or clearcut and then windrowed and planted with *Pinus elliotii* seedlings in 1972. Sections of each area (thinned or clearcut/planted) were fenced in 1977 to exclude grazing activity by deer and cattle, while the remaining portions were either open to grazing by deer only or both deer and cattle. The results showed that grazing, either by deer alone or deer and cattle in combination, did not significantly alter vascular plant cover or species diversity; however, substantial differences between the understory plant communities in the thinned versus clearcut areas were discernible. Woody plant cover in particular was higher in clearcuts and was dominated by *Pinus elliotii* and *Quercus* spp., while the thinned areas had lower percentage plant cover dominated by shrub species rather than tree seedlings. Understory species richness and diversity was consistently higher in the thinned stands than in the clearcut stands that were planted with *P. elliotii*. The authors recommend that longleaf pine forests should not be clearcut and replaced by plantation based management if ecological diversity and high quality habitat are the desired goals.

These previous studies provide insight into what can be expected in terms of general trends following the initiation of succession by a natural or anthropogenic disturbance. Perhaps the most important point taken from these studies is the influence of individual species' biological characteristics on future species composition and diversity. Knowledge of the species composition of a forest stand prior to harvest provides an invaluable resource for predicting future species composition because propagules and sprouts of these species will most likely come to dominate the sites in the future based on their ability to colonize under the specific environmental conditions that are created or changed by the disturbance. Existing knowledge of individual species' responses to

varying disturbances can also aid in understanding the resulting composition and diversity. Additionally, prior research emphasizes the need to exercise caution when using diversity measures. Such measures can provide complete information regarding the number and distribution of species within a site, but further subjective analysis must be used to interpret the dynamics of individual species with respect to their environment and each other. The complexity of ecological relationships in forests must be considered when undertaking any analysis of biodiversity.

CHAPTER II.

STUDY AREA

Physical Environment

Physiography, Geology, and Soils

The 78,000 ha Savannah River Site (SRS) is located in South Carolina within the Upper Atlantic Coastal Plain subprovince of the Atlantic Coastal Plain (ACP) physiographic province (Figure 2-1). The ACP consists of a series of old marine terraces rising gently westward to the Fall Line where the Piedmont physiographic province begins (Ware et al. 1993). The topography of the ACP varies; elevations range from sea level to slightly above 120 m (400 ft) (Quarterman and Keever 1962). Variation in relief can be attributed to erosion caused by past submergence in the Atlantic Ocean, uplift following submergence, and surficial erosion since uplift to the present. Near the Atlantic and Gulf Coasts, the lower terraces of the coastal plain are poorly dissected resulting in seasonal saturation (Christensen 2000). The province is underlain by limestone once covered by clastic sediments during the Tertiary uplift as the North American plate shifted northward from tropical to temperate latitudes (Fenneman 1938).

The SRS itself is divided further into two geographic regions: the Aiken Plateau and the coastal terraces, or “low country” (Jones et al. 1984). The study area is located on the inner, more upland, portion of the Upper ACP, corresponding to the Aiken Plateau portion of the SRS (White and Gaines 2000) (Figure 2-2). The inner terraces of the ACP exhibit more distinct dissection as more time has elapsed (since submergence) for erosion of ravines and valleys (Quarterman and Keever 1962). The inner terraces in the Aiken Plateau area generally overlie older Tertiary deposits except nearer to the Piedmont

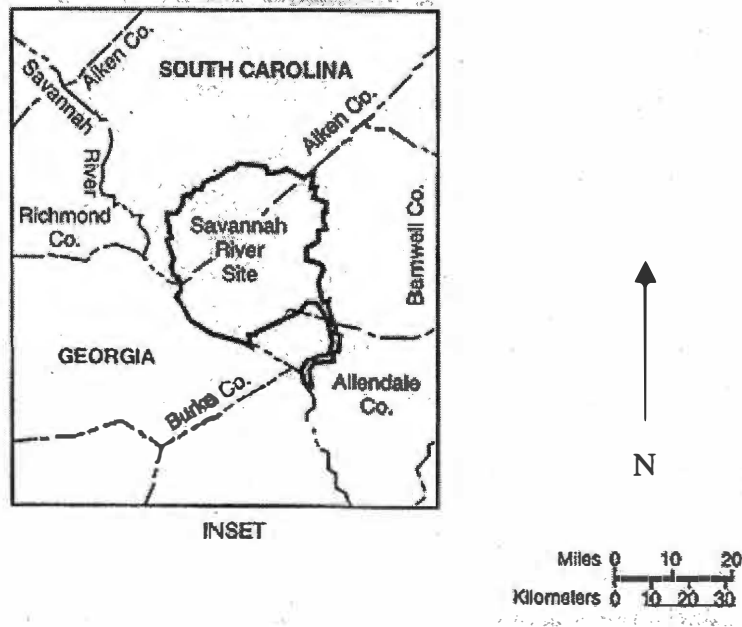
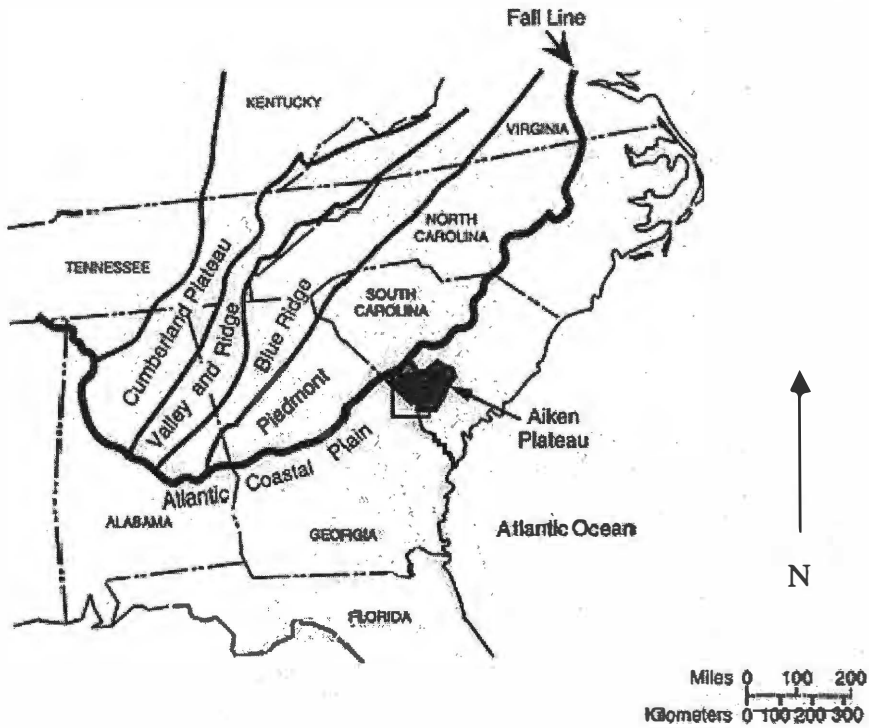


Figure 2-1. The Savannah River Site (SRS) in relation to the physiographic provinces of the southeastern United States. The SRS is located at 33° 15' N, 81° 38' W.

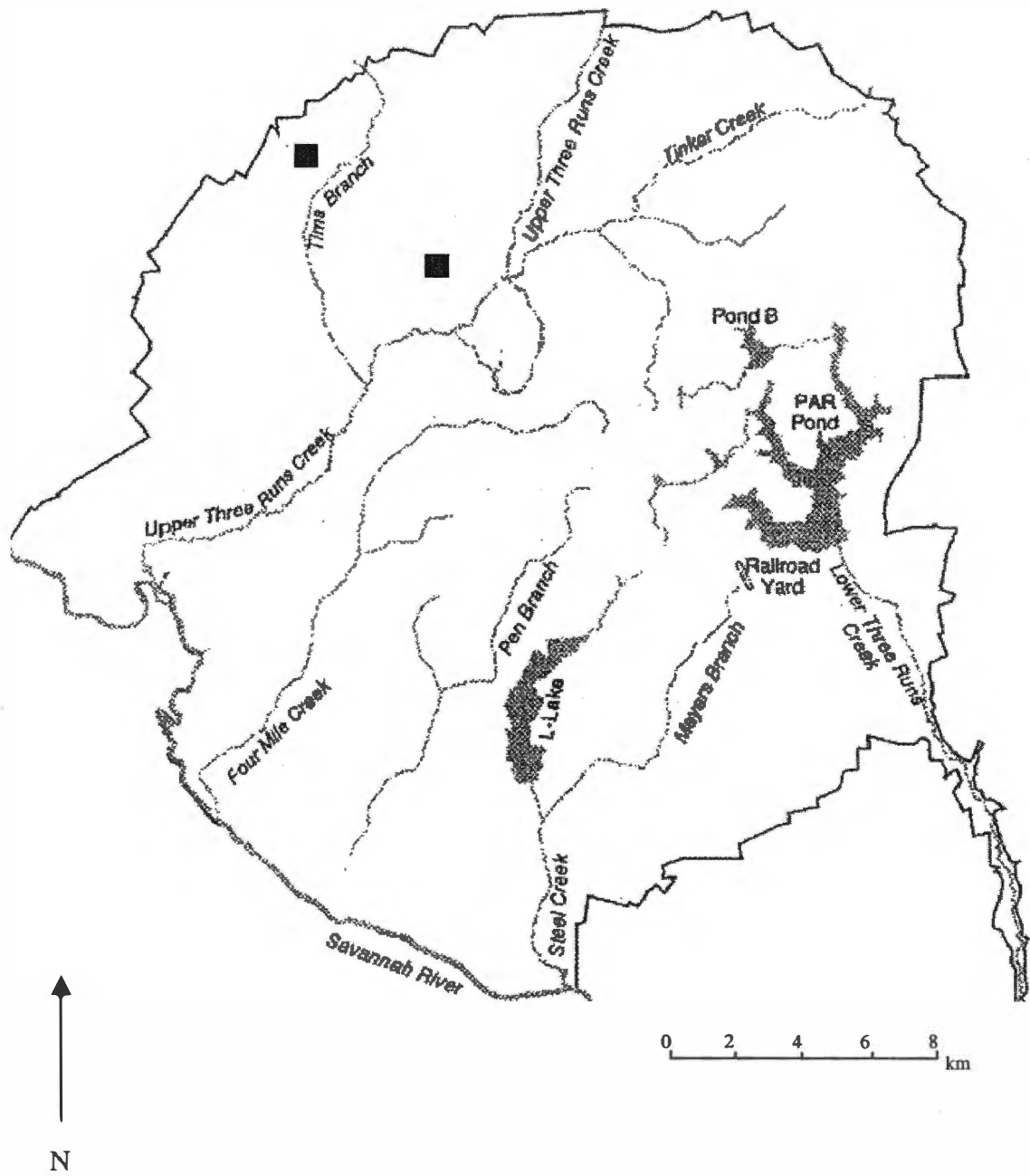


Figure 2-2. Approximate locations of the general study areas (black squares) within the Savannah River Site (SRS). The SRS is located at 33° 15' N, 81° 38' W.

where Cretaceous deposits predominate.

On the inner terraces, the predominant soils are Ultisols, and are primarily of the Udults suborder (red and yellow podzolic soils), mainly in the great groups of the Paleudults and Hapludults (Brady 1974). The prevalence of riverine and marine deposits near the Fall Line in South Carolina and Georgia contribute to soils characterized by high sand content. As a result, the Aiken Plateau portion of the SRS features upland, sandy soils of the Paleudults or Quartzipsamments suborders (Jones et al. 1984). The porosity of these soils allows swift drainage following rain, resulting in mesic to well-drained xeric habitats (Platt 1999).

Climate

The SRS falls within the humid subtropical climate zone of the southeastern United States. The ACP is warmer than the higher inland areas to the west and north, due to the moderating influences of the Atlantic Ocean and the Gulf of Mexico (Ware et al. 1993). Mean July temperatures range from 25–27° C, while mean January temperatures range from 5–16° C (Stout and Marion 1993). The average number of frost-free days ranges from 365 in southern Florida to about 250 in North Carolina (Platt 1999). The ACP has the highest precipitation within the broad Eastern Deciduous Forest Biome, averaging >122 cm per year, with the highest frequency of heavy downpours and also the most rain-free days per year (Ware et al. 1993). Weather disturbances include occasional tornadoes in the spring, while hurricanes are uncommon (Langley and Marter 1973).

Past and Present Vegetation Cover

Ware et al. (1993) present a comprehensive summary of the large body of literature that comprises efforts to classify Atlantic Coastal Plain vegetation. They summarize:

“Authors emphasizing past vegetation, present dominant vegetation, or preferred timber species for management have mapped the region as Longleaf Pine Forest (Sargent 1880), Southeastern Pine Forest (Shantz and Zon 1924), Southeastern Evergreen Forest (Braun 1950), Longleaf-Slash Pine Forest and Loblolly-Shortleaf Pine Forest (U.S. Geological Survey 1970), or fire-subclimax within the Deciduous Forest region (Weaver and Clements 1938).”

Others have emphasized the potential natural upland vegetation and have named the vegetation cover on the basis of the hardwood species that tend to dominate when fire is excluded (Ware et al. 1993). For example, Quarterman and Keever (1962) listed two successional trends of Coastal Plain forests in the absence of disturbance: (1) pines tend to be replaced by hardwoods; (2) hardwoods show an increase in total number of potential overstory species with increased age and lack of disturbance, eventually leading to shared dominance of hardwoods and pines. As a result, they developed and described the Southern Mixed Hardwood Forest Association as the ultimate dominant association over time in the absence of disturbance. The dominant and co-dominant species (nomenclature follows Radford et al. 1965) of this “post-pine” coastal plain forest consisted of: *Fagus grandifolia* Ehrh. (American beech), *Magnolia grandiflora* L. (southern magnolia), *Quercus laurifolia* Michx. (swamp laurel oak), *Quercus alba* L.

(white oak), *Liquidambar styraciflua* L. (sweetgum), *Carya tomentosa* (Poir.) Nutt. (mockernut hickory), *Quercus nigra* L. (water oak), *Quercus falcata* Michx. (southern red oak), *Carya glabra* Mill. (pignut hickory), *Nyssa sylvatica* Marsh. (blackgum), and *Ilex opaca* Ait. (American holly).

This Southern Mixed Hardwood Forest Association (Quarterman and Keever 1962) provides an accurate description of past and present vegetation in the study area at the Savannah River Site. Prior to harvest, a forest inventory of the study sites found the stands to be either pine-hardwood (stands in which 51–69% of dominant and codominant crowns are pines) or hardwood-pine (stands in which 51–69% of the dominant and codominant crowns are hardwoods) mixtures (McMinn 1989). Common tree species found within the experimental forest stands analyzed in this study include: *Quercus velutina* Lam. (black oak), *Quercus marilandica* Muenchh. (blackjack oak), *Quercus falcata*, *Quercus coccinea* Muenchh. (scarlet oak), *Quercus laevis* Walt. (turkey oak), *Quercus stellata* Wangenh. (post oak), *Quercus nigra* L. (water oak), *Quercus phellos* L. (willow oak), *Quercus alba*, *Cornus florida* L. (flowering dogwood), *Liquidambar styraciflua*, *Nyssa sylvatica*, *Carya* Nutt. (hickory), *Pinus taeda* L. (loblolly pine), *Pinus palustris* Mill. (longleaf pine), and *Prunus serotina* Ehrh. (black cherry).

Cultural History

Among eastern forests, the region considered to potentially be mixed hardwood forest is also considered one of the most disturbed (Ware et al. 1993). The longleaf pine forests and mixed pine-hardwood forests that may or may not have existed on upland sites were cleared manually before 1700 in areas conducive to settlement and near the transportation corridors of Native Americans and early European settlers, such as coastal

regions and areas along streams and rivers (Frost 1993). In addition, naval stores operations and the introduction of feral livestock as early as 1565 changed Atlantic Coastal Plain forests (Frost 1993). In the early 1700s, water-powered sawmills were introduced and larger-scale logging operations ensued along waterways (Hindle 1975). Railroad construction after the introduction of steam power in the 1830s facilitated even larger-scale logging operations and massive turpentine operations that set the stage for highly intensive management of pine forests. By the mid 1800s, feral hogs had reached very high densities on open range throughout the range of longleaf pine. After the end of the Civil War, steam-logging methods had been perfected and huge tracts of land were sold to railroad companies who then sold forested lands to logging companies. By 1880, virtually all longleaf pine forests had been removed from near streams and railroad lands and interior virgin forests were just beginning to be exploited (Frost 1993).

In parallel with logging, by the time of the Civil War, all lands optimal for agriculture were in production. By 1900, about 27% of the longleaf pine upland was listed as “improved” farmland, a category that included pastures, roads, and buildings in addition to cropland (US Census Office 1902). By 1920, fire suppression policies to protect timber were in place. Most previously unmanaged forest lands were converted to intensely managed pine plantations and remain in this status today. In 1943, the use of prescribed fire was approved by the USDA Forest Service but its use was and remains very limited on public forest land.

The lands acquired by the Department of Energy in the formation of the SRS were 34% old fields, 15% swamp and stream bottoms, and 51% mixed pine (cut-over second growth) and scrub oak (White and Gaines 2000). In 1951, the USDA Forest Service was

authorized by the Atomic Energy Commission (AEC) to manage about 65,000 hectares of the SRS. In addition, the Forest Service was also designated as a consultant to the AEC and the du Pont Company (White and Gaines 2000). Management focused on reforesting acquired farmland with the planting of 24,000 ha of *Pinus elliottii* and *Pinus palustris* by 1960. These areas were later converted to loblolly pine plantations in the 1970s using mechanical and chemical means to treat sites for planting or releasing desired trees from competition (White and Gaines 2000). Harvest of sawtimber and pulpwood using management to produce even-aged stands began in 1955 and increased when planted pines reached merchantable size after 1960 (White and Gaines 2000). The use of prescribed fire began in the early 1970s and increased during the period from 1979–81, after which air quality control issues caused the decline of its use. In 1990, prescribed fires were re-introduced to recover habitat for the red-cockaded woodpecker (*Picoides borealis* Vieill.) by restoring pine savanna through the reduction of fuel accumulation, reduction of logging slash, and discouraging the establishment of competing hardwood species (White and Gaines 2000).

CHAPTER III.

METHODS

Study Design

Framework

The original study for this project was established in 1989 as a split-split-plot randomized complete block design. In simple terms, three large areas (replicates) were chosen and treatments were randomly assigned to main plots, sub-plots, and sub-subplots separately within each replicate. This was done to ensure that each treatment combination appeared at least once in each replicate. The alternative would be a completely randomized design in which treatment combinations would be assigned randomly to sub-subplots among the three study areas, rather than dividing them into replicates. The problem with a completely randomized design would be that the replications will probably not be identical and the fact that each treatment combination may not appear in all of the three different replicates would make it impossible to separate causes of any non-uniformity in results. In this study, instead, four separate timber stands comprising three replicates were chosen. Site conditions were assumed to be relatively uniform between the replicates.

This study was designed to test the effects of two harvesting treatments, three severity levels of prescribed fire, and two pine regeneration techniques (Figure 3-1). All study areas are located in the northwest region of the SRS. The SRS is systematically divided into forest compartments that each comprise several forest stands. A forest stand can generally be defined as a reasonably homogeneous assemblage of plants that can be considered and treated as one unit (Smith et al. 1997). This study includes four stands

	Commercial Harvest Area						Silvicultural Harvest Area																					
	D			G			D			G																		
	H	L	N	H	L	N	H	L	N	H	L	N																
	↓	↓	↓	↓	↓	↓	↓	↓	↓	↓	↓	↓																
Block 1 n = 24 (Replicate 1)	P	S	P	S	P	S	P	S	P	S	P	S	P	S	P	S	P	S	P	S	P	S	P	S				
Block 2 n = 24 (Replicate 2)	P	S	P	S	P	S	P	S	P	S	P	S	P	S	P	S	P	S	P	S	P	S	P	S	P	S		
Block 3 n = 24 (Replicate 3)	P	S	P	S	P	S	P	S	P	S	P	S	P	S	P	S	P	S	P	S	P	S	P	S	P	S	P	S

Figure 3-1. Split-split-plot randomized complete block design: schematic diagram showing designated treatments. Actual spatial arrangement is pictured in Figure 3-2. Abbreviations: Main plots: D = dormant season harvest; G = growing season harvest. Subplots: H = high severity burn; L = low severity burn; N = no burn. Sub-subplots: P = planted pine regeneration; S = natural pine regeneration. The sample plots (not shown) were established within the sub-subplots.

that form three replicates. Replicates were chosen by Forest Service personnel based on the availability of hardwood-pine stands at the SRS at the time the study was established. Two harvesting treatments were used: a commercial clearcut (removal of all merchantable stems) or a silvicultural clearcut (removal of all stems >11.4 cm (4.5 in) diameter at breast height (dbh)). Initially, each replicate was divided into four treatment areas (main plots) approximately 3 ha (8 acres) in area: (1) dormant season-commercial clearcutting, (2) dormant season-silvicultural clearcutting, (3) growing season-commercial clearcutting, and (4) growing season-silvicultural clearcutting. Main plots were split into three 0.8 ha (2 acre) subplots to test the effects of three post-harvest site-preparation burn treatments: no burn, a low severity burn, or a high severity burn. Subplots were split into two 0.2–0.4 ha (0.5–1.0 acre) sub-subplots to compare pine regeneration by planting or seeding. Permanent sample plots, a total of 24 per replication, were established inside the sub-subplots after the harvest and burn treatments. The areas that were cut with the silvicultural clearcut method were not assessed for this project, leaving a total of 12 sample plots (one of each treatment combination) per replicate for a total of 36 sample plots.

The sample plots were inlaid as squares or rectangles as near to the center of the sub-subplots as possible so that the remaining area of the sub-subplots could serve as a buffer between the larger treatment areas (sub-plots and sub-subplots). Sample plots also served the purpose of standardizing the sizes, as the size and shape of the replicate areas varied. Sample plot sizes also varied slightly with stand size and location (Table A-1).

Five 13 mm (1/2 in) pvc pipes driven into the ground designated the center and four corner points of each sample plot. Sample plots were split into quadrants using the

center point as a guide. Quadrants were used as the main areas for collecting vegetation data.

Main plots, sub-plots, sub-subplots, and sample plots were established in SRS forest compartments 8 and 15; two of the replicates were in stands 38, 18A, and 18B in compartment 8, while the third replicate was established in compartment 15, stand 22 (Figure 3-2). Stands 18A and 18B in compartment 8 were considerably smaller than stands 22 and 38, and were combined to form a single replicate so that the areas of all three replicates were closer in size. The two stands are adjacent to one another but were designated as separate stands when the USFS established compartments and stands at the time of the formation of the SRS. Each replicate contains 12 sample plots for a total of 36 sample plots (Figures 3-3–3-5a and b).

Treatments

The three replicates were all harvested in 1990. Dormant season harvests were conducted in February or early March, while growing season harvests were conducted in May or June. Commercial clearcuts were conducted by commercial loggers under contract with the Savannah River Forest Station; harvesting removed all pines and hardwoods of merchantable size within each stand (Figure 3-6). These commercially harvested areas are the areas analyzed in this study.

Two fire prescriptions were applied to generate the upper and lower extremes of fire severity (as evidenced by the degree of duff consumption). High and low severity burns were conducted on July 30, 1990 and August 4, 1990 respectively. All burned sub-plots were burned using head fires and strip head fires. Head fires burn with the wind or

33 N 22'

33 N 22'

81 W 43'

81 W 39'



33 N 18'

33 N 18'

81 W 43'

81 W 39'

0 2Km

0 1Mi

Figure 3-2. Map showing the general locations of treatment areas: Compartment 8, Stand 38 (Replicate 1); Compartment 8, Stands 18A and 18B (Replicate 2); and Compartment 15, Stand 22 (Replicate 3).

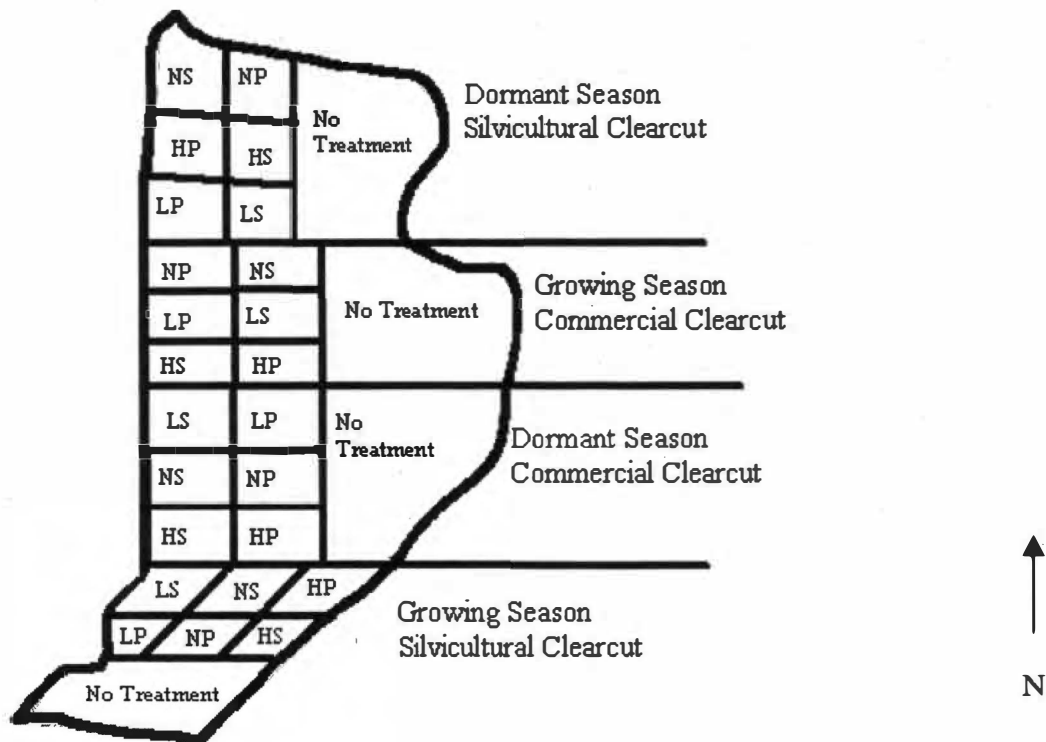


Figure 3-3. Replicate one: Compartment 8, Stand 38. NS = no burn, seeded; NP = no burn, planted; LS = low severity burn, seeded; LP = low severity burn, planted; HS = high severity burn, seeded; HP = high severity burn, planted. Approximate total treatment area = 18.5 ha (45.8 acres).

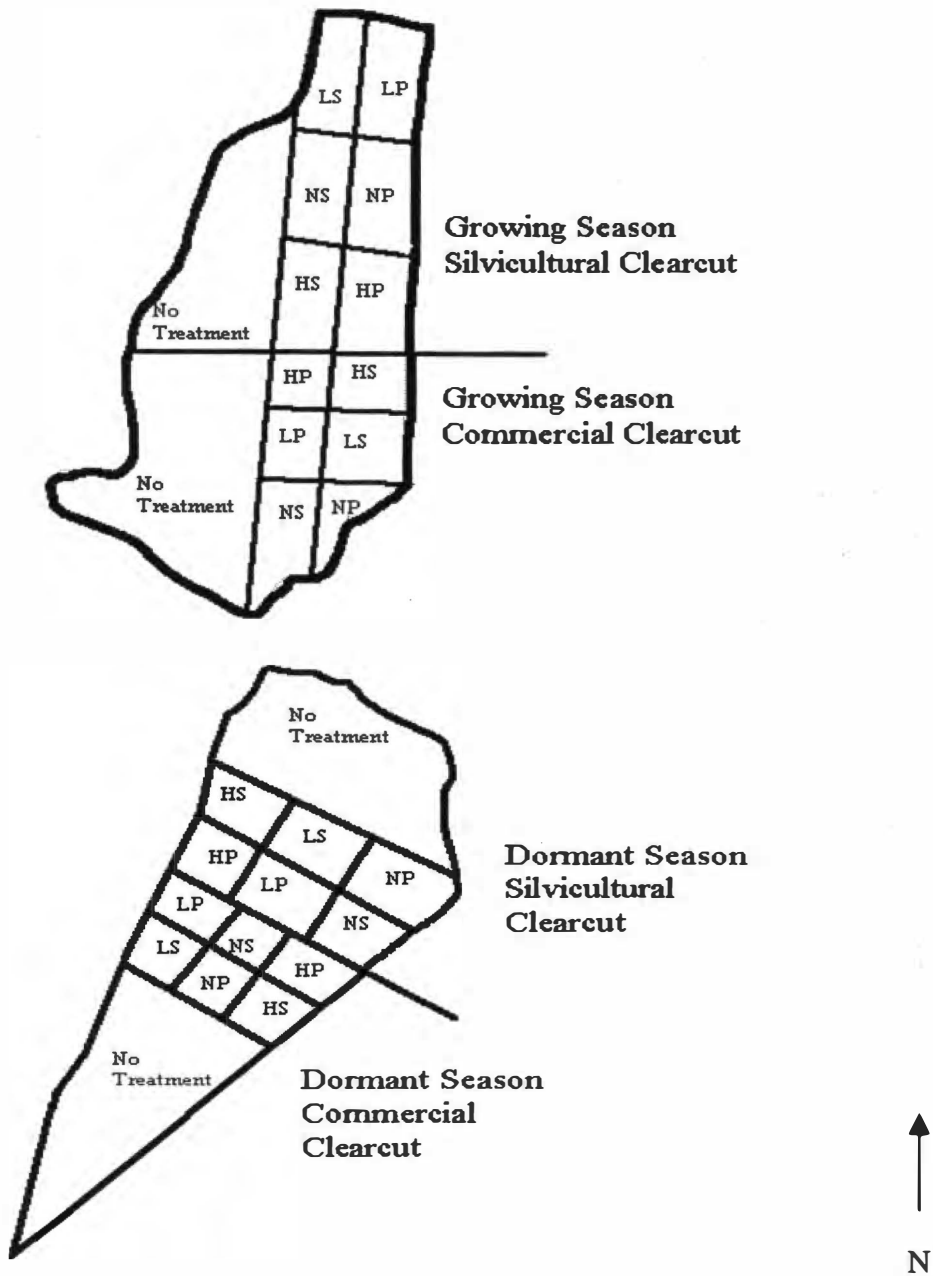


Figure 3-4. Replicate two: Compartment 8, Stand 18A (top), Stand 18B (bottom). NS = no burn, seeded; NP = no burn, planted; LS = low severity burn, seeded; LP = low severity burn, planted; HS = high severity burn, seeded; HP = high severity burn, planted. Approximate total treatment area = 20.5 ha (50.8 acres).

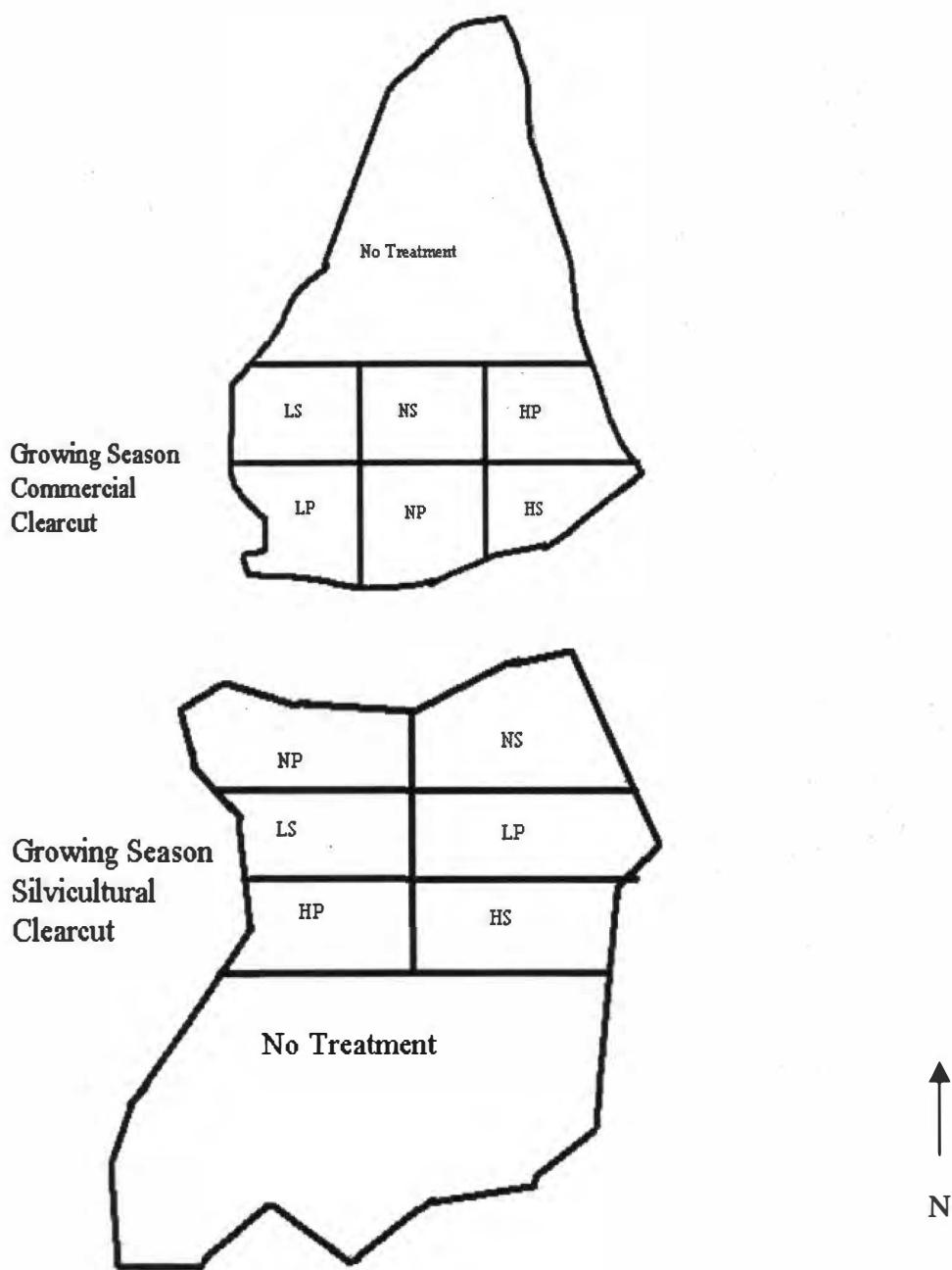


Figure 3-5a. Replicate three: Compartment 15, Stand 22 (north and middle sections). NS = no burn, seeded; NP = no burn, planted; LS = low severity burn, seeded; LP = low severity burn, planted; HS = high severity burn, seeded; HP = high severity burn, planted. Approximate total treatment area = 8.4 ha (20.8 acres).

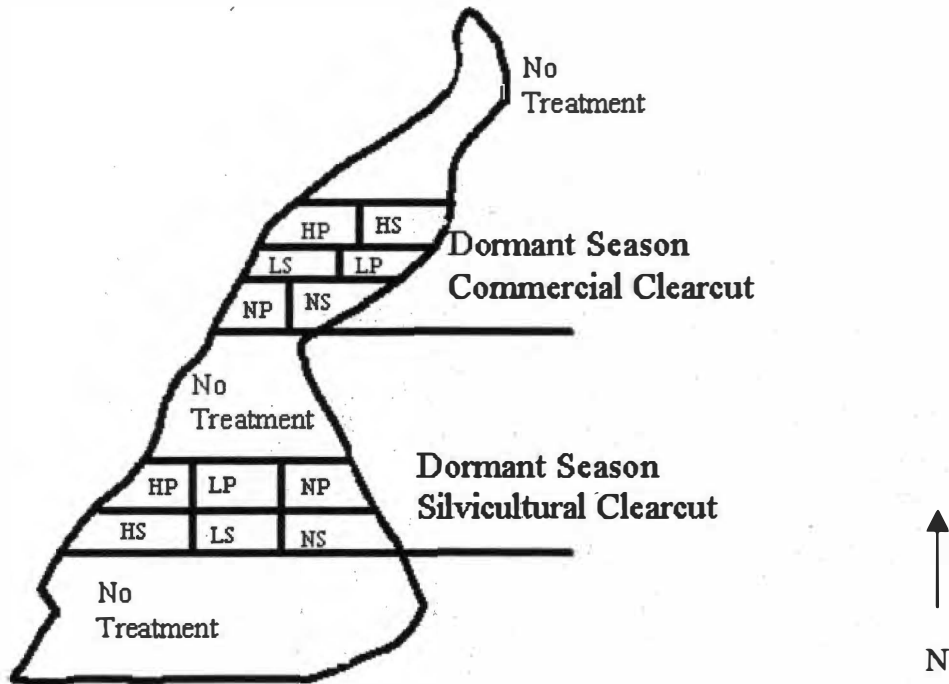


Figure 3-5b Replicate three (continued): Compartment 15, stand 22 (south section). NS = no burn, seeded; NP = no burn, planted; LS = low severity burn, seeded; LP = low severity burn, planted; HS = high severity burn, seeded; HP = high severity burn, planted. Approximate total treatment area = 20.2 ha (50.0 acres).



Figure 3-6. Commercially clearcut treatment area before site preparation or regeneration treatments.

upslope and generally have high flame lengths (indicating high intensity). Head fires are also ignited as a solid line that tends to spread faster and builds up intensity more quickly than a strip head fire or a backing fire (a fire that burns against the wind). Strip head fires (Figure 3-7) are ignited in strips next to strips that have already been ignited. Strip head fires provide better control over intensity of the fire than head fires. Fire intensity generally increases with the increase of the strip width. For both head and strip head fires, a backing fire is usually set at the opposite end of the strip to provide control by allowing for “burning out.” When fuels are fine and evenly distributed with little risk of losing control of the fire, a single head fire is preferred to a strip head fire as it reduces the number of areas of increased fire intensity that occur each time a backing fire and strip head fire burn together or burn each other out (Wade and Lunsford 1989).

The size, distribution, and moisture content of litter and debris (fuels) are used to determine the optimal time for a site preparation burn. Timelag is the drying time, under specified conditions, required for dead woody fuel to lose about 63% of the difference between its initial moisture content and its Equilibrium Moisture Content (EMC). If conditions remain unchanged, the general rule is that a fuel will reach 95% of its EMC after four timelag periods (Wade and Lunsford 1989). In this study, ten-hour timelag fuels, dead roundwood 1–3 cm (0.25–1 inch) in diameter or the top 2 cm (0.75 inch) of the litter layer, were assessed along transects within each sub-plot (burn) area. Optimal conditions for prescribed surface fires are fine, evenly distributed fuels with low moisture content.

Slash and coarse woody debris were inventoried by a Forest Service field crew immediately after harvest, and before and after burning. Transects were sampled to

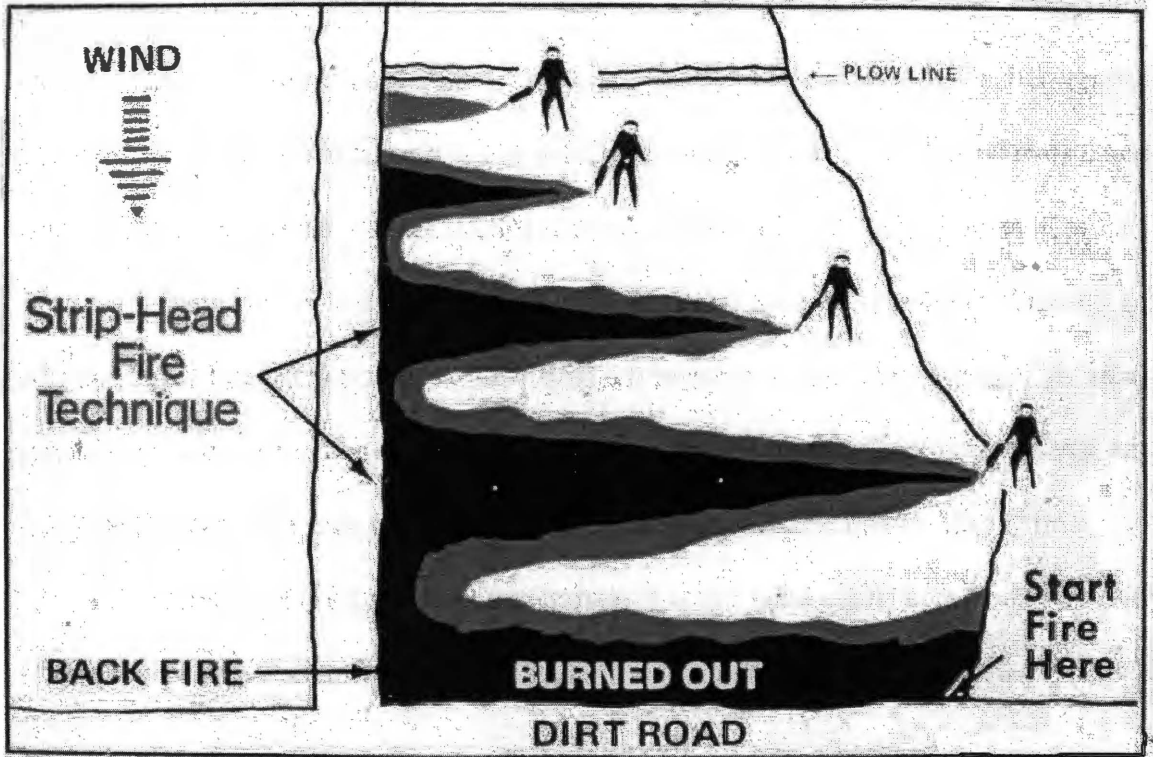


Figure 3-7. Strip-head firing technique showing strip arrangement in relation to wind direction and ignition sources (depicted by human figures) (Wade and Lunsford 1989).

estimate fuels (depth of fuels above ground and density of stems and twigs), degree of soil exposure, and consumption of duff. The measurements collected after the burn were made along the same transects as the pre-burn measurements to confirm that the burn treatments were of the desired severity. These measurements indicated that the designated sub-plots were in fact burned at the desired severity.

High-severity burns were conducted five days after a rain of 114 mm (4.5 in). This was the first precipitation within two weeks and fuels dried quickly following the event (Waldrop 1991). Moisture content of 10-hour timelag fuels ranged from 9% at the time of ignition (2:00 p.m. EDT) to 8% after burning (4:00 p.m.). Winds were from the southwest at approximately 2.2 mps (5 mph). Flame heights were estimated to range from 1.2–2.4 m (4–8 feet). The desired burn severity was achieved by using head fires only.

Low-severity burns were conducted two days after a rain of 13 mm (0.5 in). Moisture content of 10-hour timelag fuels was 12.5% at the time of ignition (10:45 a.m. EDT) and 10% after burning (2:00 p.m.). Winds were from the west at approximately 2.2–3.1 mps (5–7 mph). Flame heights were estimated to range from 0.9–2.1 m (3–7 ft). The desired burn severity was achieved using head fires in most cases, and strip head fires where more control was necessary.

Loblolly pine seedlings averaging 31 cm (1 ft) tall were planted in the plots designated for planting. Seedlings were planted during February and March of 1991, the season after the harvest and burn. Seedlings were planted with spacing of 4.6 by 4.6 m (15 by 15 ft). The plots that were not planted with seedlings were artificially seeded with *Pinus taeda* seeds using hydro-seeders at a rate of 0.56 kg per ha (0.5 lb per acre). Follow-up assessment established that seeding failed to produce any seedlings (T.

Waldrop, personal communication, August 18, 2001). As a result, all seedlings that established in later years in seeded plots are considered to have regenerated naturally rather than from seed dispersed artificially.

Data Collection Procedures

Prior to harvest in 1990, species and size class distributions of all woody vegetation were sampled in each entire sample plot by Forest Service crews. Information for each tree (species, height, dbh) was recorded on a map (stump map) for each sample plot. The mapped locations of the trees within the sample plots were estimated by the field crew. After harvest, these inventory (stump) maps were used to help differentiate sprouts regenerating directly from stumps of the harvested trees, and seedlings that happened to establish there. Species information, number and type of sprouts, and height of tallest sprout within a clump were recorded on photocopies of the original sample plot maps. Identical data were collected in March of 1994 using photocopies of the first post-harvest sample plot maps. I used these data to estimate the mean heights by species.

In October of 1991 (one growing season after harvest) and March of 1994 (three growing seasons after harvest), all regenerating “stems” (includes mature stems, sprouts, and seedlings) were identified to species when possible and tallied in the entire southwest and northeast quadrants of each of the sample plots.

In March of 2002 (eleven growing seasons after harvest), a field crew relocated the sample plots including their plot centers and four corner posts. We used measuring tapes in the field to temporarily designate the sample plot boundaries and to designate the quadrants in the field. Our assessments included tallying and identifying all stems to species when possible. We did this comprehensively for the southwest and northeast

quadrants of each sample plot, replicating the 1991/1994 methods. In addition, we estimated (or measured using a measuring stick) the heights of the tallest stems within each same-species clump other than *Pinus*. The *Pinus* species (*P. taeda* and *P. palustris*) differ from most of the hardwood species encountered in this study because they do not regenerate in clumps via prolific sprouting, but rather grow solely as individuals from seed. Therefore, we recorded each individual *Pinus* stem and estimated its height or measured it using the measuring stick.

In addition to woody species, herbaceous species data were also collected in June of 1992 and in June and August of 2002. I was able to obtain a copy of the methods used in the 1992 collection and attempted to replicate these when collecting the 2002 data. The herbaceous data were not analyzed as part of this thesis, and will be analyzed and summarized separately from the woody species data.

Three transects of approximately 28 m (93.3 ft) were traversed, through the center and along two edges, of each of the sample plots. The three transects were numbered consecutively as one, two, and three for each sample plot, with transect one always originating in the northeast corner of the sample plot and continuing in a straight line from north to south, ending near the southeast corner (sometimes passing beyond it) when the desired length was reached. Transect two traversed the center of the sample plot from the north boundary to the south boundary, and transect three originated in the northwest corner and continued in a straight line toward the southwest corner until the desired length was reached. For each of the three transects, we estimated cover for all vegetation ≤ 1.4 m (4.5 ft) in height. Any plant or plant part that crossed the transect was counted as “cover” and the length of the portion of the plant intersecting the transect was

measured to the nearest centimeter. Each plant was identified to genus and species whenever possible. Thus, for each species encountered, a list was made denoting lengths of all of the individual instances of occurrence so that these could be tallied to give a total length (cm) for each species to eventually yield a proportion of total cover relative to the length of the transect. In addition to transects, a walk-through of the sample plots was also conducted to census any species that did not cross the transects. These species were included on the overall species list but cover was not estimated.

Data Analysis

Species Density

Each of the three data sets (1991, 1994, and 2002) was compiled and analyzed separately. To calculate species densities, I summed the number of stems for each species within each measured quadrant, then divided the total stems of each species over all quadrants by the combined area of the quadrants. The area measurements of the quadrants were calculated from the original sizes of the sample plots, which were established as permanent plots with fixed areas (Table A-1). I estimated relative densities for each species by dividing the number of sprouts of each species totaled for the measured quadrants by the total number of sprouts of all kinds, to yield a proportional density of each species.

The cumulative species list, of all three data sets, includes 33 taxa (species where possible, or grouped by genus where identification proved impractical). I divided the taxa into coherent groups to simplify the interpretation of the numeric results. Species groups were adapted from those used in a study conducted by Muncy (1980) who listed species groups for the Catoosa Wildlife Management Area in Tennessee. I grouped additional

species not documented by Muncy based on general taxonomic and growth characteristics adapted from work done by Burns and Honkala (1990). *Quercus* L. (oak), *Carya* Nutt. (hickory), and *Pinus* L. (pine) species density were of special interest to this study because species in these groups were the most abundant prior to harvest and are more economically important. *Carya* species were not identified to species in any of the three years (1991, 1994, 2002) because of intrinsic variability, especially of juveniles, and interspecific hybridization, which makes them difficult to identify beyond the genus level (Hardin et al. 1996).

In addition to stem density, I also calculated mean heights for each species using data on the sample plot (stump) maps from 1991 and 1994. Because stump maps were not used in 2002, height data were recorded in the sprout and seedling tallies for that year. I used these data to extract mean heights by species and species groups. Height data should be interpreted with caution because of differences in the sizes of the areas that were sampled for height. To clarify, in 1991 and 1994, heights were collected over the area of the entire sample plot, and in 2002, heights were collected only in the two quadrants sampled for stem density. Thus, the 1991 and 1994 data provide a larger sample area to project a more accurate mean than those data collected in 2002. In addition, the average maximum height within each clump was measured, leaving the opportunity for residual trees (not harvested) to skew the data. Although I could not use these height data in statistical analyses, they provide valuable information on growth dynamics of pines and hardwoods.

Diversity Indices

Several numeric indices are available for measuring species diversity; Magurran (1988) provides an excellent summary. I chose the following three diversity indices: (1) richness, (2) evenness or equitability, and (3) the Shannon-Wiener (Shannon) diversity index. I chose these indices because they are the most commonly used and accepted diversity indices in current vegetation diversity literature.

Richness is simply the number of woody species observed in each sample plot. I chose richness because it is a simple indicator of species numbers as succession progresses over time. Species richness alone, however, is most often not adequate to detect subtle changes that may occur in the diversity of a community. When species richness is relatively stable or unchanged over time or across an environmental gradient, there might be shifts in the distribution of abundance among existing species. Other diversity indices were developed specifically to detect these types of changes.

The Shannon index was developed to incorporate both species richness and equitability into a single number. The Shannon index assumes that individuals are randomly sampled from an indefinitely large population (Pielou 1969). The index also assumes that all species within the community are represented in the sample. The Shannon index (H') is derived with the equation:

$$H' = -\sum_{i=1}^s p_i \ln p_i$$

where the quantity p_i is the proportion of individuals or the abundance of the i th species expressed as a proportion of total cover, s is the total number of species, and \ln is the natural log. The true p_i value is rarely known and is thus estimated as n/N , the maximum

likelihood of the estimator (Pielou 1969). Any base logarithm can be used, provided it is kept constant when comparing samples. Values of the Shannon index usually lie between 1.5 and 3.5, with an exceptional case exceeding 4.5. The higher the value of H' , the higher the diversity of the sample.

Evenness, or equitability, represents the proportions or relative abundances of species over an area. The calculation of evenness was described by Huston (1994) as the ratio of a composite index to the theoretical value of that index if all species in the sample were equally abundant. Basically, the index represents how each of the species that are present is represented within a sampled area. Values of the index range from 0 to 1. If all of the species are found in equal proportions, the evenness value will be closer to one than if a particular species dominates.

The equitability or evenness index (J) can now be calculated directly from the Shannon index values with the equation:

$$J = \frac{H'}{H'_{\max}} = \frac{-\sum_{i=1}^s p_i \ln p_i}{\ln s}$$

Where H'_{\max} is the maximum value of H' , s is the number of species, p_i is the proportion of individuals of the i th species or the abundance of the i th species expressed as a proportion of total cover, and \ln is the natural log. The value of J is higher when species are more even in distribution within the sample. I chose to use J in addition to H' because it aids in demonstrating the manner in which both evenness and species richness contribute to the Shannon index in studied samples.

I used the Pisces Diversity program (Henderson and Seaby 1999) to calculate diversity. I entered relative density data into a spreadsheet to generate diversity values for each sample plot, treating each of the three data sets (representing three different ages of the stands) separately. It is very important that the same diversity measures be used consistently in the confines of a given study, to provide results that are interpretable and realistic. Species richness, the Shannon index, and the equitability index all meet this criterion as calculated for the individual plots of this study.

Analysis of Variance (ANOVA)

Analysis of variance (ANOVA) was determined to be the most useful in this case because the procedure can be used to compare means between two or more samples. The analysis of variance procedure requires that the sample data are normally distributed and thus assumed to be representative of the population at large. I used the SAS computer software package (SAS 1999) to ascertain whether the data were normally distributed using the NORMAL procedure in SAS. Once normality was confirmed, I conducted ANOVA between samples within each of the three sampled years (1991, 1994, and 2002).

I conducted two separate sets of analyses using two sets of data as dependent variables: species group density data in one analysis and diversity indices in a second analysis. The ANOVA procedure was used to test means between treatments within each year and also tested for differences in means with interactions of the treatments. Results with a probability of ≤ 0.05 (95 percent confidence level) were considered significant.

CHAPTER IV.

RESULTS

General Information

Sprouts, seedlings, and mature stems were counted in each of the three years that data were recorded (322,576 in 1991; 362,375 in 1994; and 163,931 in 2002). Although I did not take part in the 1991 and 1994 surveys, I processed all of the paper data files for these surveys. I found only a few minor discrepancies in species coding and tally numbers. Because of the age of the raw data files and predictable turnover of field personnel, I interpreted the few typographical errors to the best of my judgement. Each of the three data files was processed and interpreted separately. I first summarize the species composition of the three stands prior to harvest. I then summarize the general trends of the aggregate species densities and average heights of species over the three sampling periods. I next compare species densities and proportions with management treatments and analyze these results statistically. Finally, I summarize the results of the diversity index calculations and statistical comparisons of those measurements.

Terminology

I use the term “stems” throughout this section to refer to all sprouts, seedlings, or mature stems that were tallied. Sprouts and seedlings are obviously at a different stage of development than saplings or mature stems. However, because the number of living, residual stems was limited and the data did not distinguish them, all sprouts, seedlings, and stems were grouped together and treated equally in analyses. The tallies of sprouts and seedlings are very important because they indicate both the recovery and resiliency of particular species. In addition, they serve as an indicator of the potential of a species to

colonize or maintain itself within, and possibly dominate, a site. I use the term “density” to express the number of stems per hectare. “Relative density” is the percentage of stems of a particular taxon relative to the total number of stems in a sample plot or multiple plots of a particular treatment. “Frequency” is simply the number of sample plots in which a species occurred. The maximum frequency in any given year is 36, as 36 sample plots were tallied in each of the three years. Each species or genus was assigned to a larger “species group” (Burns and Honkala 1990): *Carya* Nutt. (hickory), *Quercus* L. (oak), *Pinus* L. (pine), the group containing the other potential overstory woody dicots (referred to as OPO hereafter), and understory woody dicots (referred to as UWD hereafter) (Table A-2).

Burn Severity

Tables 4-1 and 4-2 illustrate the results of the site preparation burn treatments. The results indicate that differences in levels of burn severity were achieved. The high severity fires burned a larger percentage of the treatment area and consumed the floor in a larger percentage of area. In addition, the high severity fires produced larger patches that were burned leading to a smaller number of patches per ha relative to the low severity fires that had smaller but more numerous patches of burned area.

Species Density

Stands Prior to Harvest

Based on the pre-harvest inventory maps for each sample plot, I calculated relative density for each stand and for the total of the three stands (Figure 4-1). The stands were primarily dominated by species of the genus *Carya* and *Quercus*. Less than

Table 4-1. Percent of each treatment area burned at various severity levels¹.

Season of harvest	Burn severity prescription	Unburned	Low severity burn	High severity burn	High severity burn- no vegetation
Dormant season	Low severity	68.8	21.2	9.5	0.5
	High severity	17.2	30.2	46.6	6.0
Growing season	Low severity	39.2	38.4	22.4	0.0
	High severity	11.1	46.1	42.4	0.4

¹ Low severity = presence of charred organic debris with no soil exposure.

High severity = forest floor entirely consumed, exposing mineral soil.

High severity-no vegetation = mineral soil exposed by burning and no seed germination.

Table 4-2. Average size and number of burned and unburned patches within each burn severity treatment area.

Season of harvest	Burn severity prescription	Burned areas		Unburned areas	
		Patch size (m ²)	Number per ha	Patch size (m ²)	Number per ha
Dormant season	Low severity	46.1	67	126.3	54
	High severity	175.4	47	24.9	69
Growing season	Low severity	47.7	121	37.8	111
	High severity	256.6	35	10.2	109

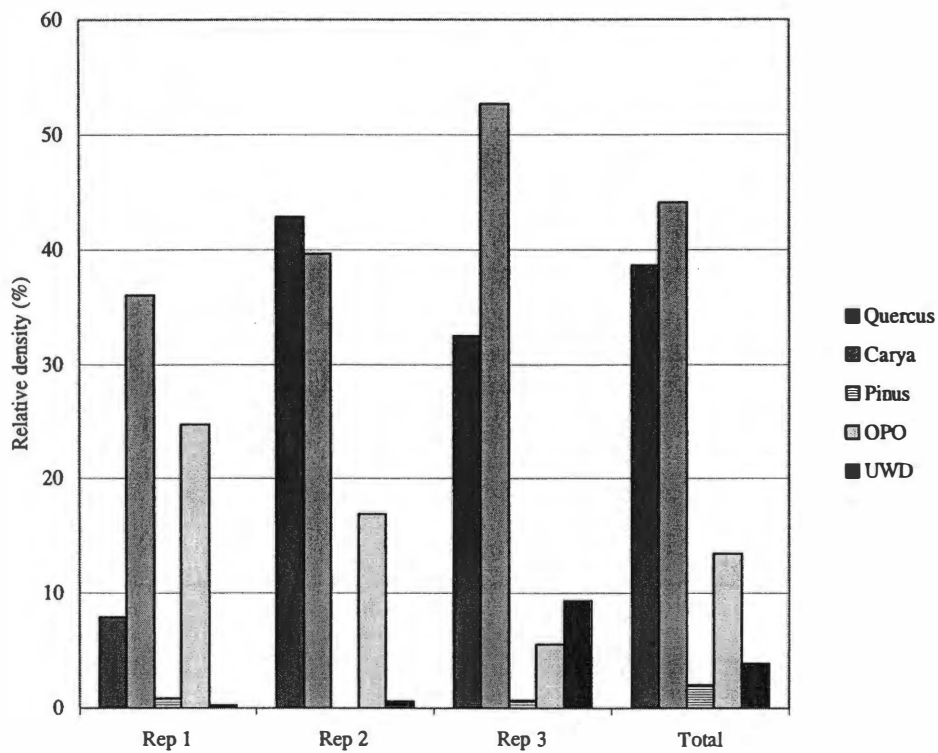


Figure 4-1. Relative densities of each species group prior to harvest for the three replicates calculated from pre-harvest inventory maps. OPO = Other potential overstory species, UWD = understory woody dicots (see Tables 4-1 and A-2 for details on species groups).

1% of trees in all stands were composed of the *Pinus* group (primarily *Pinus taeda*) prior to harvest. The previous relative densities of adult trees can lend a considerable amount of contextual information to interpretation of the results. A cautionary note should be mentioned. The UWD relative density values may have been undercounted in the pre-harvest inventory because the main goal then was to assess the larger and more abundant stems that dominated the treatment areas and were likely to be harvested. Although the UWD group may be underrepresented, it is likely that they were still not anywhere near the relative densities calculated in the years after the harvest.

General Trends (1991-2002)

For all three years, the UWD group had the highest relative density values (Figure 4-2, Table 4-3). These values remained stable for the duration of the study, comprising 55–62% of all stems. The main species components of the UWD group were *Vaccinium arboreum* Marsh. (sparkleberry) and *Rhus* spp. (sumac), while *Cornus* spp. (dogwood) was a secondary component.

The *Quercus* and *Carya* groups ranked behind UWD in relative density values over time. The relative density values of the two groups were stable over the course of the study (~12–13%). The main contributor to the *Quercus* group was consistently *Q. falcata* Michx.

Pinus showed a steady increase in relative density over the 11 year period. *Pinus taeda*, the species that was planted, was found in consistently higher densities than all other pines, as expected.

Relative density values among species members of OPO remained relatively stable over the course of the study. *Prunus serotina* Ehrh., *Nyssa sylvatica*

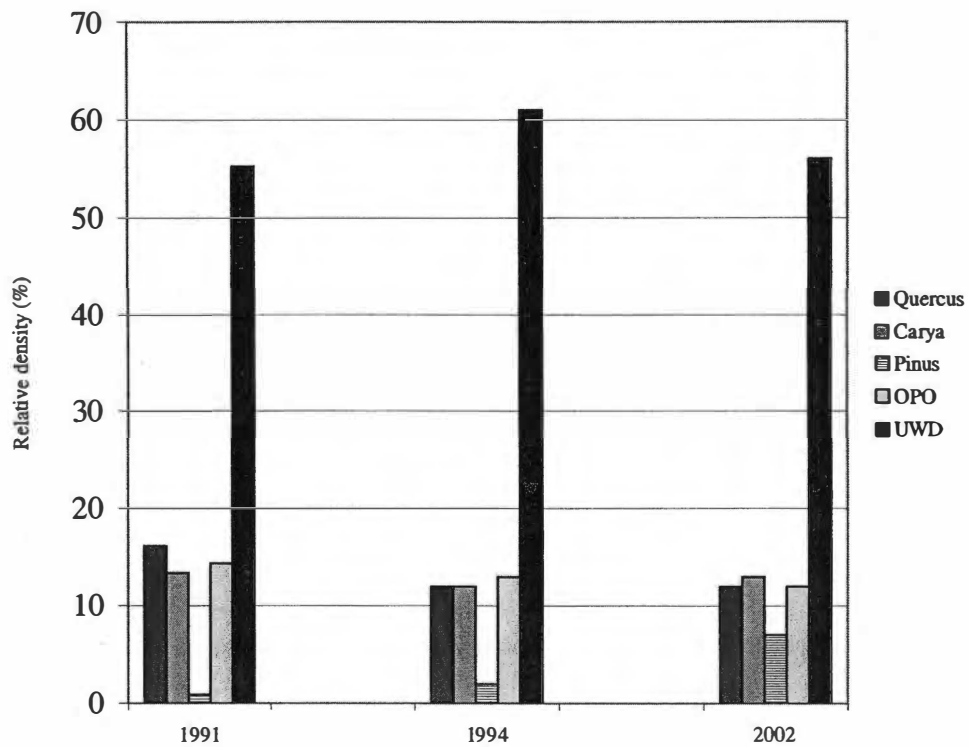


Figure 4-2. Relative density (proportion of total stems/ha) of species groups by year of data collection. OPO = Other potential overstory woody dicots, UWD = understory woody dicots (see Table A-2 for constituents of species groups).

Table 4-3. Occurrence data for each identified taxon for each sample year, listed by species group. Stems = stems/ha; relative density = proportion of total stems/ha expressed as percent; frequency = number of plots where the species occurred (n=36 for each year). Subsequent analyses are based on species group data.

Species Group	Species	1991			1994			2002		
		Stems	Relative Density	Frequency	Stems	Relative Density	Frequency	Stems	Relative Density	Frequency
<i>Quercus</i> L.	<i>Quercus alba</i> L.	141	4	28	9	<1	11	2	<1	3
	<i>Quercus coccinea</i> Muenchh.	23	1	11	7	<1	12	0	<1	1
	<i>Quercus falcata</i> Michx.	248	7	35	163	4	32	47	4	30
	<i>Quercus laevis</i> Walt.	0	0	0	1	<1	2	48	3	26
	<i>Quercus marilandica</i> Muenchh.	26	1	17	33	1	13	20	2	20
	<i>Quercus nigra</i> L.	11	<1	15	53	1	26	49	3	32
	<i>Quercus stellata</i> Wangenh.	85	2	28	150	4	31	14	1	22
	<i>Quercus velutina</i> Lam.	35	1	20	36	1	19	6	<1	8
	Total	569	16		452	11		186	12	
<i>Carya</i> Nutt.	<i>Carya</i> spp.	459	13	36	528	12	36	215	13	36
Other potential overstory (OPO)	<i>Diospyros virginiana</i> L.	47	1	29	32	1	20	10	1	27
	<i>Fraxinus</i> spp.	1	<1	2	0	0	0	0	0	0
	<i>Liquidambar styraciflua</i> L.	98	3	10	146	3	11	31	2	13
	<i>Nyssa sylvatica</i> Marsh.	308	7	26	210	5	25	53	3	30
	<i>Prunus serotina</i> Ehrh.	87	3	31	172	4	33	88	6	34
	<i>Ulmus alata</i> Michx.	13	<1	7	9	<1	6	7	<1	6
	Total	554	14		569	13		189	12	

Table 4-3 Continued.

Species Group	Species	1991			1994			2002		
		Stems	Relative Density	Frequency	Stems	Relative Density	Frequency	Stems	Relative Density	Frequency
<i>Pinus</i> L.										
	<i>Pinus taeda</i> L.	24	1	30	60	2	24	88	6	36
	<i>Pinus palustris</i> Mill.	2	<1	1	13	1	7	10	1	12
	Total	26	1		73	3		98	7	
Understory woody dicots (UWD)										
	<i>Aralia spinosa</i> L.	0	0	0	0	<1	1	0	0	0
	<i>Baccharis halimifolia</i> L.	0	0	0	0	0	0	0	<1	1
	<i>Celtis</i> spp.	2	<1	3	0	0	0	1	<1	1
	<i>Cornus</i> spp.	156	4	9	186	4	7	53	3	9
	<i>Crataegus</i> spp.	44	1	20	83	2	28	52	3	25
	<i>Ilex opaca</i> Ait.	19	<1	10	32	1	15	25	1	19
	<i>Ilex vomitoria</i> Ait.	0	0	0	0	0	0	4	<1	3
	<i>Ligustrum</i> spp.	0	0	0	2	<1	1	2	<1	1
	<i>Myrica cerifera</i> L.	0	0	0	0	0	0	7	1	9
	<i>Prunus virginiana</i> L.	0	0	0	0	0	0	3	<1	5
	<i>Rhus</i> spp.	528	15	36	692	19	35	123	8	35
	<i>Sassafras albidum</i> (Nutt.) Nees	108	3	27	23	1	21	5	<1	9
	<i>Tilia americana</i> var. <i>caroliniana</i> (Miller) Castigl.	9	<1	5	39	1	2	0	0	0
	<i>Vaccinium arboreum</i> Marsh.	1156	32	35	1397	34	36	846	37	35
	<i>Viburnum</i> spp.	0	0	0	0	0	0	28	2	8
	Total	2022	55		2454	62		1149	55	
Unknown	Unknown spp.	0	0	0	4	1	2	5	1	5

Marsh., and *Liquidambar styraciflua* L. were the major components of this group. The species with the highest relative density shifted over time from *Nyssa sylvatica* to *Prunus serotina*, although the differences in relative density between the two were small throughout (~3%).

Mean heights (Figure 4-3) add dimension to the stem density data. The height data, although lacking precision, play an important role by enabling structural visualization of the stands to illustrate the dynamics of species dominance and competition over time. The tallest stems were most likely residuals, and tended to raise the average heights in their respective species/genus groups. Over time, the UWD group was surpassed in height by the four overstory groups, as expected. *Pinus* exceeded all of the other four groups in height by 2002, a pattern commonly encountered in studies of succession (Burns and Honkala 1990).

Stands in 1991 (Age 1 year): Management Treatments

The responses to treatments were most obvious in the 1991 results, as very little time had passed between the application of the treatments and the survey. The season of harvest had an obvious effect on species density (Figure 4-4, Table A-3). Differences in the relative density of *Quercus*, *Carya*, OPO, and UWD groups were statistically significant for the season of harvest (Table 4-4).

Level of burn severity had distinct but less obvious effects on species densities (Figure 4-5). Nonetheless, results from the ANOVA did not indicate significance in any case between the severity of burn and the relative density of any of the five species groups in 1991.

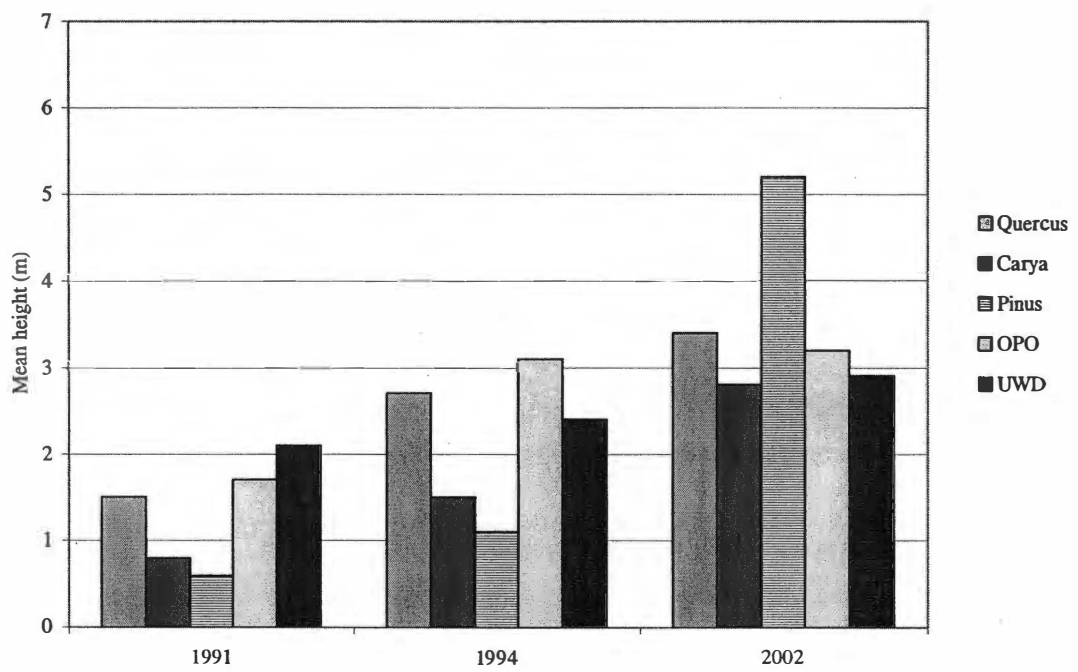


Figure 4-3. Mean heights (m) of each group by sampling year. OPO = Other potential overstory woody dicots, UWD = understory woody dicots (see Table A-2 for constituents of species groups).

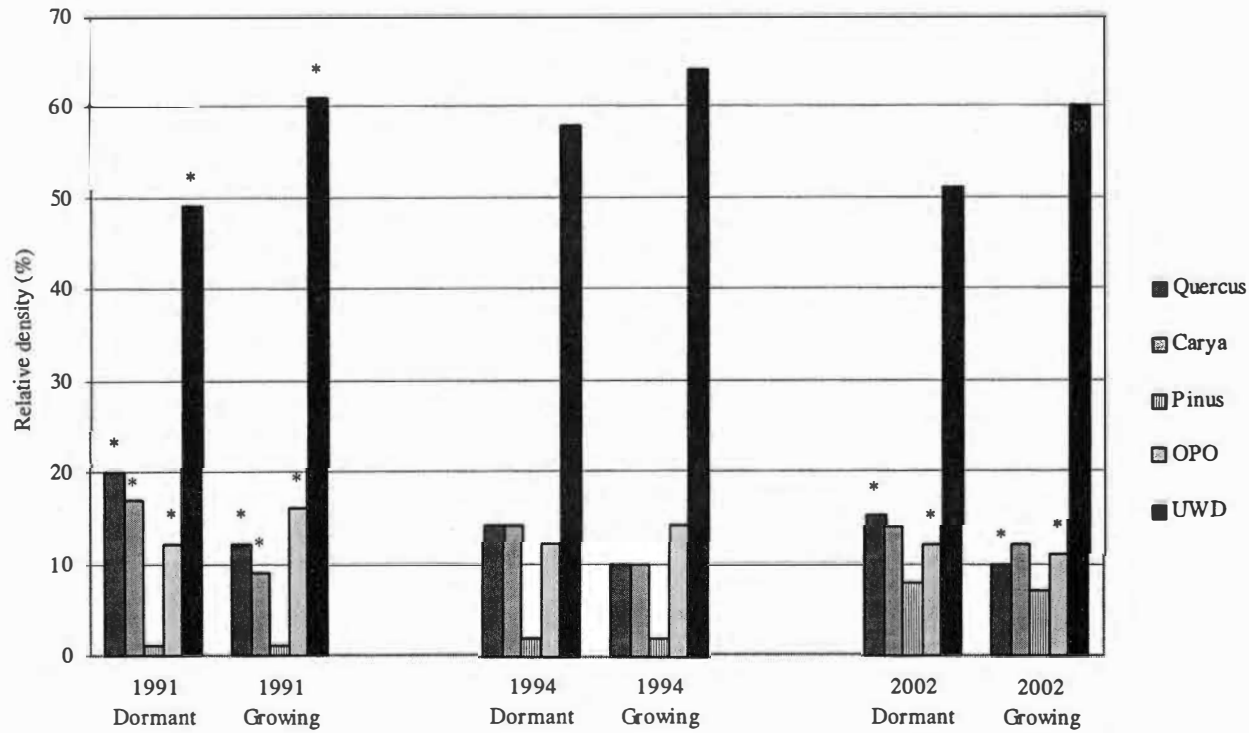


Figure 4-4. Season of harvest results. Relative density (proportion of total stems/ha) for each species group. Bars within each year with an * are significant within their species group between treatments.

Table 4-4. Results of ANOVA comparing relative density (by species group) and management treatments. An * denotes significance at the 0.05 level, ns = not significant. Only results that were statistically significant are shown. See Table A-8 for full results.

Year		<i>Quercus</i>		<i>Carya</i>		<i>Pinus</i>		OPO		UWD	
		F	Pr > F	F	Pr > F	F	Pr > F	F	Pr > F	F	Pr > F
1991	Season	4.59	0.046 *	5.70	0.028 *	ns	ns	4.72	0.043 *	3.08	0.010 *
	Season*Burn	ns	ns	ns	ns	ns	ns	ns	ns	0.38	0.020 *
	Pine regeneration	ns	ns	ns	ns	7.12	0.010 *	ns	ns	ns	ns
1994	Burn	ns	ns	4.98	0.019 *	ns	ns	ns	ns	ns	ns
2002	Season	7.31	0.015 *	ns	ns	ns	ns	7.31	0.015 *	ns	ns
	Pine regeneration	ns	ns	ns	ns	11.69	0.002 *	ns	ns	ns	ns

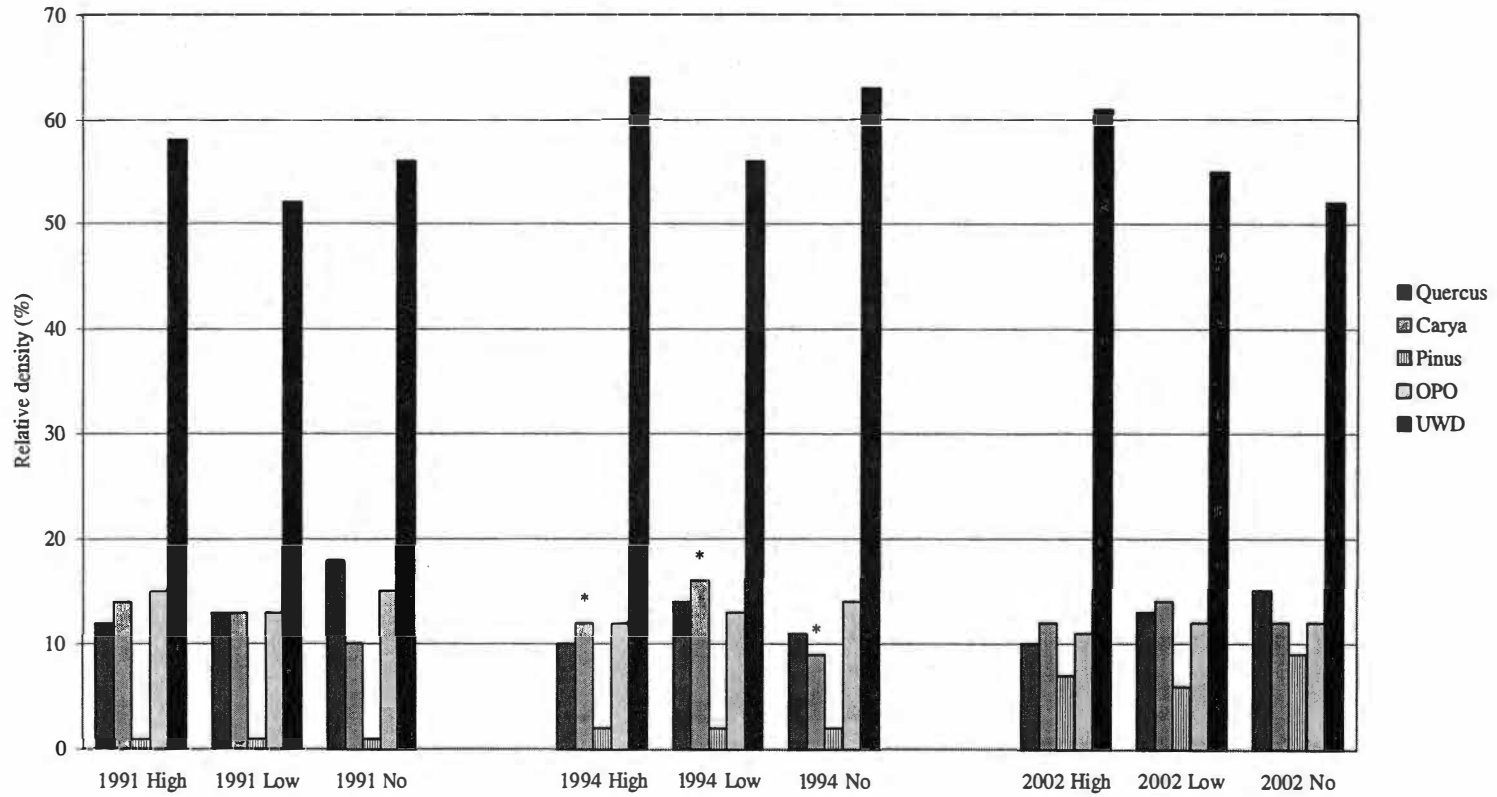


Figure 4-5. Burn severity treatment results. Relative density (proportion of total stems/ha) for each species group. Bars within each year with an * are significant within their species group between treatments.

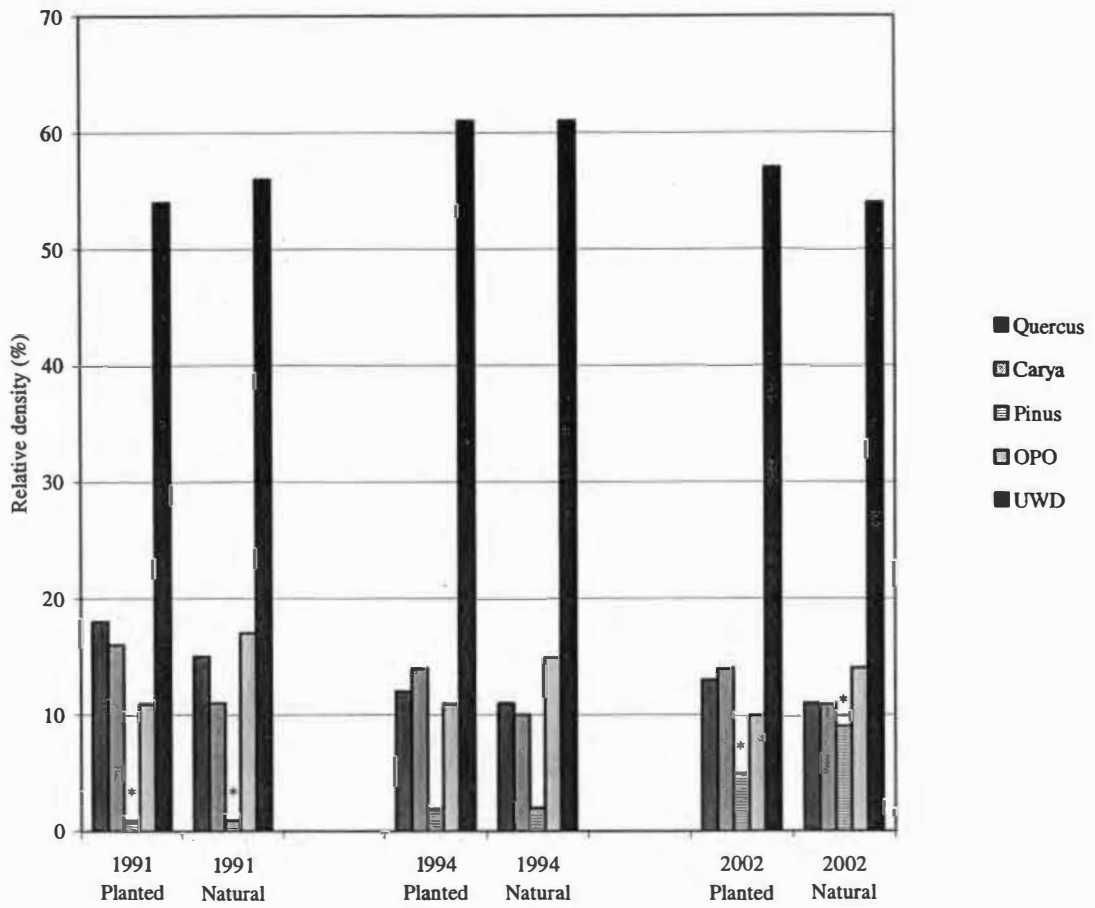


Figure 4-6. Pine regeneration treatment results. Relative density (proportion of total stems/ha) for each species group. Bars within each year with an * are significant within their species group between treatments.

At age one year, densities appeared similar regardless of pine regeneration technique (Figure 4-6). The results emphasize the extremely low overall density and relative density values of the *Pinus* group.

I was especially interested in finding out if the burn severity or pine regeneration treatments had an additive effect on relative density of the species groups when coupled with season of harvest. The results indicated a significant relationship for UWD density when season of harvest and burn severity were combined (Table 4-4).

Stands in 1994 (Age 3 years): Management Treatments

By age three years, differences between the relative densities of the four groups became less pronounced (Figure 4-4). For the season of harvest treatments, relative densities were distributed about the same as in the 1991 data and patterns found in 1991 were still weakly discernible (Table A-8). ANOVA tests between the season of harvest and relative density values of the five species groups did not indicate any significant relationships.

In 1994, the burn severity treatments yielded patterns similar to the 1991 results (Figure 4-5). In the OPO category, *Prunus serotina* and *Liquidambar styraciflua* showed large increases in density by age three, especially in the low severity burn category, thus adding to the high relative density of that group in plots of this burn severity treatment. The UWD group showed higher density values in the no burn plots due to greater density of *Vaccinium arboreum* stems in all of these plots, the reasons for which are unclear. *Carya* density was significantly different between the burn treatments in 1994 (Table 4-4) being higher in the plots that were burned at low severity.

When categorized by pine regeneration treatment, the 1994 results are nearly identical for each species group in the two categories; however, the *Pinus* group itself started to show a slightly higher density value in the plots that were allowed to regenerate naturally.

Stands in 2002 (Age 11): Management Treatments

The 2002 results illustrate that for each of the three treatments, differences between treatments of relative density values of species groups had become much smaller among treatments. Season of harvest resurfaces as a significant factor in term of density of both *Quercus* and OPO stems. This suggests that season of harvest may have effects on tree species composition up to and beyond 11 years following harvest. In addition, the density of *Pinus* stems was significantly different between the two pine regeneration treatments, with higher densities in the naturally regenerated plots.

Diversity

Tables 4-5 through 4-8 summarize species richness, Shannon index, and equitability values by site preparation treatment for each sample year. Mean richness values and equitability values were nearly equal between the three sampling periods, yielding equally similar Shannon diversity values (Table 4-5). When the mean diversity data were organized by season of harvest, all three values were consistently (although only slightly) higher in the dormant season harvest areas than in the areas harvested during the growing season (Table 4-6). When burn severity was considered, mean values

Table 4-5. Mean diversity index values for each year of data.

Year	Richness ¹	Shannon Index (H) ²	Equitability (J) ³
1991	11	1.79	0.51
1994	11	1.70	0.48
2002	12	1.81	0.51

¹Richness = average number of species.

²Shannon index values range from 1.5 (low diversity) to 4.5 (high diversity).

³Equitability (J): $0 > J \leq 1.0$, higher values indicate more equitable representation of species.

Table 4-6. Mean diversity index values separated by season of harvest.

Year	Season	Richness ¹	Shannon index (H) ²	Equitability (J) ³
1991	Dormant	12 ^a	1.88 ^a	0.53 ^a
	Growing	10 ^b	1.69 ^b	0.48 ^b
1994	Dormant	11 ^a	1.75 ^a	0.50 ^a
	Growing	10 ^b	1.66 ^a	0.47 ^a
2002	Dormant	13 ^a	1.96 ^a	0.56 ^a
	Growing	11 ^b	1.66 ^b	0.47 ^b

¹Richness = average number of species.

²Shannon index values range from 1.5 (low diversity) to 4.5 (high diversity).

³Equitability (J): $0 > J \leq 1.0$, higher values indicate more equitable representation of species.

⁴Means within a column and year followed by the same letter are not significantly different at the 0.05 level.

Table 4-7. Mean diversity index values separated by burn severity.

Year	Burn	Richness ¹	Shannon index (H') ²	Equitability (J) ³
1991	High	11 ^{b4}	1.63 ^b	0.46 ^b
	Low	11 ^b	1.79 ^b	0.51 ^b
	No	13 ^a	1.94 ^a	0.55 ^a
1994	High	11 ^a	1.64 ^a	0.47 ^a
	Low	10 ^a	1.72 ^a	0.49 ^a
	No	12 ^a	1.76 ^a	0.50 ^a
2002	High	12 ^a	1.73 ^a	0.49 ^a
	Low	12 ^a	1.86 ^a	0.53 ^a
	No	13 ^a	1.85 ^a	0.52 ^a

¹ Richness = average number of species.

² Shannon index values range from 1.5 (low diversity) to 4.5 (high diversity).

³ Equitability (J): $0 > J \leq 1.0$, higher values indicate more equitable representation of species.

⁴ Means within a column and year followed by the same letter are not significantly different at the 0.05 level.

Table 4-8. Mean diversity index values separated by pine regeneration.

Year	Pine Regeneration	Richness ¹	Shannon index (H') ²	Equitability (J) ³
1991	Natural	12 ^{a4}	1.79 ^a	0.51 ^a
	Planted	11 ^a	1.79 ^a	0.51 ^a
1994	Natural	11 ^a	1.73 ^a	0.49 ^a
	Planted	10 ^a	1.68 ^a	0.48 ^a
2002	Natural	12 ^a	1.83 ^a	0.52 ^a
	Planted	13 ^a	1.79 ^a	0.51 ^a

¹ Richness = average number of species.

² Shannon index values range from 1.5 (low diversity) to 4.5 (high diversity).

³ Equitability (J): $0 > J \leq 1.0$, higher values indicate more equitable representation of species.

⁴ Means within a column and year followed by the same letter are not significantly different at the 0.05 level.

were slightly higher for all three diversity indices in the areas that were not burned (Table 4-7). Table 4-8 shows the mean diversity indices between the two pine regeneration treatments. The results of each index were nearly equal within each year that was sampled.

I conducted ANOVA tests to compare treatments using all three mean index values, separately for each of the three data sets (1991, 1994, 2002). The full results of these analyses are presented in Table A-9; significant results are presented in Table 4-9. ANOVA of species diversity indices combined with all possible treatment interactions indicates a significant response to season of harvest and to burn treatment for all three of the diversity indices at one year of age. At age three years, a significant relationship only existed between richness and season of harvest, indicating that the number of species differed, but equitability of species was similar in the two treatments. Although the results were statistically significant, they may not be meaningful because, in this case, richness values differed by only one species.

At age 11 years, however, all three diversity indices showed a significant response to the season of harvest. Combinations of the secondary treatments with season of harvest yielded no significant results.

Table 4-9. Significant ANOVA results for mean values of diversity indices. An * indicates significance at the 0.05 level, ns = not significant.

Year		Richness ¹		Shannon index (H') ²		Equitability (J) ³	
		F	Pr >F	F	Pr >F	F	Pr >F
1991	Season	9.82	0.006 *	5.38	0.032 *	5.38	0.032*
	Burn	4.55	0.025 *	4.72	0.022 *	4.72	0.022*
1994	Season	4.24	0.054 *	ns	ns	ns	ns
2002	Season	9.96	0.007 *	5.20	0.038 *	5.20	0.038*

¹ Richness = average number of species.

² Shannon index values range from 1.5 (low diversity) to 4.5 (high diversity).

³ Equitability (J): $0 > J \leq 1.0$, higher values indicate more equitable representation of species.

CHAPTER V.

DISCUSSION

Major Findings

Over the 11 years covered in this study, diversity was not affected by the combinations of management treatments. This demonstrates that the ability of this ecosystem to recover from the inflicted disturbances, in a relatively short amount of time, is very strong. However, the sample plots and treatment areas are adjacent to one another. Thus, it is possible that characteristics of certain treatment areas might have had an effect on neighboring areas (exposed to different treatment combinations); the possibility of such effects were noted, but no attempt was made to measure them or include them in the statistical analyses. Similarly, nearby forest stands might have influenced results by contributing unequally as seed sources for the sample plots.

Although the replicates were assumed to be independent and uniform in their environmental conditions at the time this study was established, significant differences between species densities and diversity indices among replicates proved this was not necessarily the case. The results of the statistical analyses (Tables A-6 and A-7) indicate possible spatial variability between sites. This possible site variability leads to the inference that, in many cases, local environmental factors may have affected species composition and biodiversity as much as, if not more than, the management treatments themselves.

Specifically, mean stem numbers of particular species groups significantly differed between replicates in both 1994 and 2002. In addition, all three of the diversity indices differed significantly (ANOVA results) in different replicates in both 1991 and

1994. This suggests that site conditions probably stayed consistent after the treatments to support the pre-harvest species. Further, individuals that were damaged yet capable of sprouting had an advantage over those species that depended on seed dispersal from outside of the site (i.e., chance).

Furthermore, the pre-harvest density data resemble the corresponding post-harvest density data when the replicates are considered separately. This emphasizes the influence of the biological legacies of pre-harvest species on the future species compositions of harvested sites. Each site has unique local environmental and biological characteristics that were sustained in some degree through even the most severe harvest disturbances.

Although I could not statistically correlate the height data (Figure 4-3) with the different treatment combinations, the overall trend is quite clear. Mean maximum heights of all species groups increased over time. Between the ages of 3 and 11, heights increased considerably for all four of the potential overstory groups (*Pinus*, *Carya*, *Quercus*, and OPO). As the canopy closes, smaller members of shade-intolerant species will die and existing larger individuals will vie for space within the canopy. *Pinus* and the other groups will continue to experience dynamic competitive interactions as their relative heights continue to fluctuate based on the availability of and competition for resources.

Research Questions Revisited

Question 1: Do post-harvest species composition and diversity of woody species differ as a result of season of harvest?

Although season of harvest seemed mainly to have a strong immediate effect, these effects likely contributed in turn to similar but much weaker patterns in the later sampling periods. Age one results (1991) indicate that season of harvest had a major

effect on the densities of all four non-pine species groups. Even after eleven years, significant differences were detectable between the two seasons of harvest in the *Quercus* and OPO groups, indicating the lasting influence of season of harvest by its effect on initial interspecies competition (as taxa reacted to the disturbance), and the eventual population structures of the woody species of the sample plots. However, this long-term pattern is likely reflecting the success of *Carya* and *Quercus* in the short term, between ages one and three.

Quercus and *Carya* have advantages when it comes to site disturbance, as they are able to reproduce vegetatively via vigorous sprouting, whereas species in the *Pinus* group (in this case) are able to reproduce only via seed germination, and rarely produce sprouts in response to damage (Smith et al. 1997). The season of harvest can have a strong effect depending on the biological properties of the affected species. When sites are harvested in the dormant season, species capable of vegetative reproduction are able to grow vigorously in the next growing season due to high amounts of carbohydrate reserves that are stored in roots during the dormant season (McMinn 1992). A harvest, as a disturbance, facilitates intense hardwood growth via sprouting from roots and stumps holding the energy reserves. Conversely, a harvest in the growing season (in this case, the early-mid growing season following the first growth flush in most hardwood species) leads to an overall reduction in stump and root carbohydrate reserves as not enough time has elapsed (in the growing season) to accumulate photosynthate to replenish those reserves.

The 1991 results reflect this pattern. Higher relative densities of the *Carya* and *Quercus* groups were observed on the sites harvested in the dormant season. However,

the UWD group (which also includes many vegetative reproducers) had higher mean relative densities on sites that underwent the growing season harvest. A plausible explanation is that most of the *Quercus* and *Carya* trees were harvested and their root systems were damaged more than those of the smaller UWD species that were probably not harvested and therefore sustained less injury. This could have resulted in the higher numbers of *Carya* and *Quercus* sprouts in the dormant season harvest areas by their ability to regenerate from stored energy reserves, while the less-damaged understory group fared better in the growing season harvested areas due to less competition from *Carya* and *Quercus* species that had difficulty resprouting from damaged root systems with lower energy reserves.

Pinus density was very low in both of the season of harvest treatments. *Pinus* drop their seeds in fall, and harvest in the dormant season helps germination by exposing the mineral soil. However, in this case there were very low numbers of *Pinus* stems in dormant season plots, indicating that very little, if any, seeds were on the plots at the time of harvest. A growing season harvest would likely occur after seeds have had a chance to germinate, but would also prove destructive to seedlings at such a small growth stage. It is for this reason that pines are usually planted in hardwood stands after harvest, because they have little chance of becoming a large component of mixed forests otherwise. The results suggest this to be the case, especially the very low values of *Pinus* stems in 1991.

The analyses of the differences in the variances of the three diversity indices emphasized the effects of season of harvest on diversity (Table A-9). Higher richness values in the dormant season plots are exclusively due to higher numbers of non-pine species (OPO, *Quercus*, *Carya*, and UWD) found in the dormant season plots as all but

two of the species encountered (*Pinus taeda* and *Pinus palustris*) are non-pine species. As mentioned above, many of the woody species are generally more successful on sites that have been harvested in the dormant season because of their ability to sprout vigorously following dormant season disturbance.

Question 2: Is post-harvest species composition and diversity of woody species affected by surface burning following clearcutting?

Although species group stem densities did not differ significantly between burn treatments (with the exception of *Carya* in 1994), some interesting patterns emerged within each species group that clarified the significant differences in diversity among burn treatments. Higher numbers of *Quercus*, *Prunus serotina*, and *Sassafras albidum* on low severity or unburned sites after one growing season accentuate the negative effects of burning on the regeneration of these species. The high density of *Rhus* spp. in the high burn severity plots was expected based on its ability to resprout prolifically following fire (Burns and Honkala 1990). In addition, although *Sassafras albidum* had low values of relative density, its presence is of interest as it is known to be extremely sensitive to damage from fire (Burns and Honkala 1990). The higher density of *S. albidum* stems in the no and low severity burn severity treatments is corroborating evidence that the desired levels of burn severity were achieved. Prescribed fire is known to have detrimental effects on coppice sprouting hardwood species (*Quercus*, *Carya*, and members of the OPO group and UWD group) than on non-sprouting softwood species such as pines, which usually have thicker bark (at mature stages) with better insulating qualities. Likewise, hardwood seedlings and sprouts are likely to be killed by fire, while

stumps, roots, and larger stems often resprout after damage from burning. *Pinus taeda* (both planted and natural) and *Pinus palustris* seedlings are not resistant to fire at such a young stage and were not likely to survive the burns. Therefore, the highest numbers of naturally seeded pine stems were expected to be found on the no burn plots.

The significant values of the results for diversity in 1991 (Table 4-7) show that burning had the greatest effect on diversity after one growing season, when recolonization and adjustment to post-disturbance site conditions were just beginning. The sites that were not burned favored more species and yielded a more equal distribution of species than the burned sites. The fact that significant differences were only found in the 1991 data is logical; many species were likely recolonizing and no particular species had had enough time to dominate on the sites with lower levels of disturbance.

Question 3: Do stands that were planted with *Pinus taeda* seedlings exhibit differences in woody species composition and diversity compared to those where pines were allowed to regenerate naturally?

The mechanical agitation of the soil during clearcutting and the burning of the litter layer can favor germination of many previously dormant and buried seeds, but can also be detrimental to young seedlings. Although the *P. taeda* seedlings were given a head start at the onset of succession, the rapid flush of hardwood stems arising from the existing seedbank and residual, undamaged tree parts was not significantly altered by competition from the planted *P. taeda* seedlings or volunteer pine seedlings. This is evident in the similar values for species diversity indices in each of the pine regeneration treatments. Therefore, the presence of planted *P. taeda* seedlings at the initiation of

succession did not affect the number or evenness of non-pine species found at the three periods sampled because the seedlings were likely killed in the site preparation burn treatments. This is true because the number of planted pines was so small that they did not affect these measures of diversity. However, this is not to say that the pines that exist now will not likely dominate in size in the near future (as the height data suggest; Figure 4-3).

In unplanted plots, *Pinus* depend on seeds (rather than sprouting) to reproduce. Therefore, existing seed stock in the sample plots, outside seed sources such as adjacent pine forests, and the planting of *P. taeda* seedlings (which can begin producing seeds from ages 5–10 and prolifically by age 25), likely all played a role in the increased abundance of pine stems after eleven years. Outside seed sources (from adjacent pine stands or adjacent planted sample plots), or existing dormant seeds in the duff layer, may also have contributed to the increase in *Pinus* densities in the naturally regenerated plots by 2002 (age 11). These findings suggest the need for artificial pine regeneration by planting if *Pinus* species are a desired goal of management. In the absence of planted pines, dependence on seed germination alone solely relies on chance occurrence from existing seedbanks or adjacent sources and the success of pines cannot be guaranteed, especially on sites similar to these that are dominated by hardwood species prior to harvest.

Disturbance

The results of this study show that hardwood-pine communities in the SRS and most likely elsewhere in the ACP are resilient to disturbances associated with the management techniques that were investigated. In the absence of intense hardwood

reduction techniques such as prolonged herbicide application, hardwood species will continue to persist where they have persisted in the past. This is obvious in the relatively short time frame (11 years) within which the effects of management showed little remaining effect on species composition and diversity. I can further conclude from the evidence that antecedent conditions, a combination of specific site conditions leading toward the presence and success of certain species, can have a strong influence on post-disturbance composition and diversity under the influence of limited types of management disturbances similar to those employed here.

Components of the initial floristics composition model proposed by Egler (1954) seem applicable to the developmental processes that take place in harvested stands. In Egler's model, virtually all members of the entire successional sequence are present following the abandonment of agricultural land. The differences in the current study occur because succession began after clearcutting, burning, and planting, which agitate the soil, favoring germination of previously dormant and/or buried seeds and sprouting from living plant parts that survived both the harvest and burn disturbances. These conditions correlate with Egler's view that suggests that initial conditions might fix the course of succession, in contrast to other studies that suggest compositional differentiation increases with succession (e.g., Matthews 1979; Pineda et al. 1981). Likewise, Felix et al. (1983) found that trends in richness and evenness of cut-over forests (secondary successional forests) differed from old-field succession models due to a high diversity of seeds and sprouts at the start of succession.

Margalef (1963) proposed that, in pioneer communities, convergence occurred from an initial, highly random composition toward a largely deterministic climax created

through increased species fidelity to site conditions over time with increased competition and decreased niche breadth. Although this study does not assess pioneer communities, it supported the idea that species fidelity to sites that existed prior to harvest (and additional disturbance) plays a deterministic role in the initiation phase of secondary succession.

In terms of the site preparation burning, Arthur et al. (1998) found that a single fire promoted sprouting, whereas repeated fire increased the mortality of understory stems and sprouts and reduced subsequent sprouting over time. In the current study, repeated fires might have had a more noticeable effect on species composition and diversity due to increased heterogeneity of the burned areas, increasing the likelihood of specialization and survival of particular species (fire-tolerant or intolerant) depending on the treatment applied. However, microsite variations were not documented as part of the data collection process.

This study was especially valuable in terms of its timespan. The effects of the applied management disturbances were not strongly evident after 11 years of recovery but season of harvest had effects at ages one and three years that set up the successional sequence in terms of species composition and diversity. Although species densities and diversity did not differ significantly at 11 years after the various combinations of disturbance and pine regeneration, the significant differences shortly after the disturbances most likely had a major influence on the future characteristics of the sample plots. Although the study design limited the sizes of the disturbed areas and introduced complexity in terms of surrounding environmental and biological influences, it also proved a valuable lesson in site selection for this type of experiment because the results

suggest that very specific (micro)site conditions (not measured in this study) may have a greater effect on composition and diversity than the disturbances themselves.

Management Considerations

The results of this research are beneficial to managers with a variety of perspectives and goals. If the goal of management is for aesthetic benefit and wildlife habitat, the results of this study show that clearcutting does not cause short or long-term (11 years in this case) changes in woody species composition or diversity after the harvest. The same species that were present prior to harvest remained present after the harvest.

One of the economic goals of pine-hardwood management is to add a pine component to stands normally dominated by hardwoods in order to add value to managed stands. The treatments in this study show that both planted pines, and possibly outside sources of pine seed, can lead to a strong pine component in future stands, while at the same time keeping a strong and diverse hardwood component. However, over time, all three treatments proved to be relatively equal in their effects on species regeneration, composition, and diversity. Thus, the benefits of conducting mixed species management can profit managers that have both economic and ecological interests, because expensive site preparation treatments are largely unnecessary when the goal is to create mixed hardwood-pine stands that retain pre-disturbance species composition and diversity.

CHAPTER VI.

CONCLUSION

Timber harvesting and site preparation management techniques are used extensively on all forested lands, public and private, in the Southeast. These types of practices are often the most intense, and by far the most frequent, of disturbance that forests in this region experience. As the demand for wood and wood products increases, more intense and more frequent management disturbances will result. Because southeastern forests must serve multiple purposes (e.g., economic, recreational, aesthetic), it is imperative that management practices be examined with regard to their effects on the ecosystems that they modify.

This study was established in the Savannah River Site to learn how Atlantic Coastal Plain hardwood-pine forests react and recover from harvesting and site preparation treatments. My results demonstrate that single management disturbances (burn and season of harvest) have strong short term effects on woody species composition and diversity. While I did not detect strong effects after 11 years, the effects in the short term might have affected the post-disturbance woody species composition and diversity and these effects may continue to influence the species composition and diversity of the study area.

The results indicated that along with the initial response from disturbance, woody species composition prior to disturbance was probably a strong influence on the post-disturbance composition of the study sites. In addition, physical site characteristics that might have supported specific assemblages of species would have persisted in the post-harvest sample plots. Prior species composition and physical site characteristics might

also interact: pre-harvest vegetation cover might have modified soils and the microclimate of specific sites such that these conditions might have been able to persist even after the most intense management disturbances were applied.

In terms of management, this research shows that clearcutting can be conducted in these types of forests without eliminating or changing the composition of species. Thus pine-hardwood mixtures can be grown successfully at low cost because very little site preparation is needed to establish mixed stands. If more economic value is desired, more intense site preparation techniques would be needed to introduce and maintain a stronger pine component that will add increased value to these types of stands.

The results, in aggregate, show that the forest management treatments used in this experiment do not have a lasting effect on woody species composition and diversity in Atlantic Coastal Plain hardwood-pine forests. The canopy plant assemblages present prior to the harvest (primarily *Quercus* and *Carya*) were resilient and thus able to recover relatively quickly from the management treatment disturbances. This is not to say that diversity and species densities on these types of sites (upland hardwood-pine forests) would not have fluctuated more with repeated, and/or more intense management practices. More work should be done that focuses on comparing other, more intense management strategies common to these types of sites, especially the intense practices used for conversion from hardwood-pine sites to pine plantations, which is likely more reflective of the current and the future management regime in the region.

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APPENDIX

Table A-1. Plot identification codes, terminology, and sizes in ft² and m². Compartment/stand refers to SRS designations (See Figure 3-2). For the meanings of sample plot, total quadrant, and total area sampled see Chapter II.

Plot name	Season of harvest	Burn severity	Pine regeneration treatment	Replicate	Compartment/stand	Sample plot area m ² (ft ²)	Total quadrant area m ² (ft ²)	Total area sampled m ² (ft ²)
GCHP1	Growing	High	Planted	1	8/38	297 (6400)	74 (1600)	149 (3200)
GCHS1	Growing	High	Natural	1	8/38	297 (6400)	74 (1600)	149 (3200)
GCLP1	Growing	Low	Planted	1	8/38	297 (6400)	74 (1600)	149 (3200)
GCLS1	Growing	Low	Natural	1	8/38	297 (6400)	74 (1600)	149 (3200)
GCNP1	Growing	No	Planted	1	8/38	297 (6400)	74 (1600)	149 (3200)
GCNS1	Growing	No	Natural	1	8/38	297 (6400)	74 (1600)	149 (3200)
DCHP1	Dormant	High	Planted	1	8/38	297 (6400)	74 (1600)	149 (3200)
DCHS1	Dormant	High	Natural	1	8/38	297 (6400)	74 (1600)	149 (3200)
DCLP1	Dormant	Low	Planted	1	8/38	297 (6400)	74 (1600)	149 (3200)
DCLS1	Dormant	Low	Natural	1	8/38	297 (6400)	74 (1600)	149 (3200)
DCNP1	Dormant	No	Planted	1	8/38	297 (6400)	74 (1600)	149 (3200)
DCNS1	Dormant	No	Natural	1	8/38	297 (6400)	74 (1600)	149 (3200)
GCHP2	Growing	High	Planted	2	8/18A & 18B	402 (8649)	100 (2162)	201 (4324)
GCHS2	Growing	High	Natural	2	8/18A & 18B	402 (8649)	100 (2162)	201 (4324)
GCLP2	Growing	Low	Planted	2	8/18A & 18B	402 (8649)	100 (2162)	201 (4324)
GCLS2	Growing	Low	Natural	2	8/18A & 18B	402 (8649)	100 (2162)	201 (4324)
GCNP2	Growing	No	Planted	2	8/18A & 18B	149 (1600)	37 (400)	74 (800)
GCNS2	Growing	No	Natural	2	8/18A & 18B	297 (6400)	74 (1600)	149 (3200)
DCHP2	Dormant	High	Planted	2	8/18A & 18B	297 (6400)	74 (1600)	149 (3200)
DCHS2	Dormant	High	Natural	2	8/18A & 18B	297 (6400)	74 (1600)	149 (3200)
DCLP2	Dormant	Low	Planted	2	8/18A & 18B	297 (6400)	74 (1600)	149 (3200)
DCLS2	Dormant	Low	Natural	2	8/18A & 18B	297 (6400)	74 (1600)	149 (3200)
DCNP2	Dormant	No	Planted	2	8/18A & 18B	297 (6400)	74 (1600)	149 (3200)
DCNS2	Dormant	No	Natural	2	8/18A & 18B	297 (6400)	74 (1600)	149 (3200)

Table A-1 continued.

Plot name	Season of harvest	Burn severity	Pine regeneration treatment	Replicate	Compartment and stand location	Sample plot area m ² (ft ²)	Total quadrant area m ² (ft ²)	Total area sampled m ² (ft ²)
GCHP3	Growing	High	Planted	3	15/22	297 (6400)	74 (1600)	149 (3200)
GCHS3	Growing	High	Natural	3	15/22	297 (6400)	74 (1600)	149 (3200)
GCLP3	Growing	Low	Planted	3	15/22	297 (6400)	74 (1600)	149 (3200)
GCLS3	Growing	Low	Natural	3	15/22	297 (6400)	74 (1600)	149 (3200)
GCNP3	Growing	No	Planted	3	15/22	297 (6400)	74 (1600)	149 (3200)
GCNS3	Growing	No	Natural	3	15/22	297 (6400)	74 (1600)	149 (3200)
DCHP3	Dormant	High	Planted	3	15/22	297 (6400)	74 (1600)	149 (3200)
DCHS3	Dormant	High	Natural	3	15/22	167 (3600)	42 (900)	83 (1800)
DCLP3	Dormant	Low	Planted	3	15/22	167 (3600)	42 (900)	83 (1800)
DCLS3	Dormant	Low	Natural	3	15/22	167 (3600)	42 (900)	83 (1800)
DCNP3	Dormant	No	Planted	3	15/22	297 (6400)	74 (1600)	149 (3200)
DCNS3	Dormant	No	Natural	3	15/22	297 (6400)	74 (1600)	149 (3200)

Table A-2. Complete list of woody species observed. Nomenclature follows Radford et al. (1965). Species groups are adapted from Muncy (1980) and Burns and Honkala (1990).

Species Group	Scientific name	Common name
<i>Quercus</i> L.	<i>Quercus alba</i> L.	White oak
	<i>Quercus coccinea</i> Muenchh.	Scarlet oak
	<i>Quercus falcata</i> Michx.	Southern red oak
	<i>Quercus laevis</i> Walt.	Turkey oak
	<i>Quercus marilandica</i> Muenchh.	Black jack oak
	<i>Quercus nigra</i> L.	Water oak
	<i>Quercus stellata</i> Wangenh.	Post oak
	<i>Quercus velutina</i> Lam.	Black oak
<i>Carya</i> Nutt.	<i>Carya</i> spp.	Undifferentiated Hickories
Other potential overstory woody dicots (OPO)	<i>Diospyros virginiana</i> L.	Common persimmon
	<i>Fraxinus</i> L.	Ash
	<i>Liquidambar styraciflua</i> L.	Sweetgum
	<i>Nyssa sylvatica</i> Marsh.	Blackgum
	<i>Prunus serotina</i> Ehrh.	Black cherry
	<i>Ulmus alata</i> Michx.	Winged elm
<i>Pinus</i> L.	<i>Pinus taeda</i> L.	Loblolly pine
	<i>Pinus palustris</i> Mill.	Longleaf pine
Understory woody dicots (UWD)	<i>Aralia spinosa</i> L.	Devil's walking stick
	<i>Baccharis halimifolia</i> L.	Baccharis
	<i>Celtis</i> L.	Hackberry
	<i>Cornus</i> L.	Dogwood
	<i>Crataegus</i> L.	Hawthorn
	<i>Ilex opaca</i> Ait.	American holly
	<i>Ilex vomitoria</i> Ait.	Yaupon
	<i>Ligustrum</i> L.	Privet
	<i>Myrica cerifera</i> L.	Wax myrtle
	<i>Prunus virginiana</i> L.	Plum
	<i>Rhus</i> L.	Sumac
	<i>Sassafras albidum</i> (Nutt.) Nees	Sassafras
	<i>Tilia americana</i> var. <i>caroliniana</i> (Miller) Castigl.	Carolina basswood
	<i>Vaccinium arboreum</i> Marsh.	Sparkleberry
<i>Viburnum</i> L.	Viburnum	
Unknowns	Unknown spp.	Unknowns

Table A-3. Mean number of stems per hectare by season of harvest.

Species group	Species	1991		1994		2002	
		Dormant	Growing	Dormant	Growing	Dormant	Growing
<i>Quercus</i> L.	<i>Quercus laevis</i> Walt.	0	0	1	1	61	35
	<i>Quercus nigra</i> L.	14	7	56	50	57	42
	<i>Quercus coccinea</i> Muenchh.	14	31	12	3	0	1
	<i>Quercus marilandica</i> Muenchh.	42	11	51	14	27	14
	<i>Quercus velutina</i> Lam.	52	18	46	25	9	2
	<i>Quercus stellata</i> Wangenh.	106	64	165	134	18	10
	<i>Quercus alba</i> L.	142	139	12	5	2	2
	<i>Quercus falcata</i> Michx.	292	205	189	137	56	38
	Total	662	475	533	368	230	144
<i>Carya</i> Nutt.	<i>Carya</i> spp.	557	361	590	465	263	168
Other potential overstory species (OPO)	<i>Fraxinus</i> L.	0	2	0	0	0	0
	<i>Ulmus alata</i> Michx.	19	7	17	1	13	1
	<i>Diospyros virginiana</i> L.	56	38	46	17	12	9
	<i>Liquidambar styraciflua</i> L.	96	100	110	182	22	39
	<i>Nyssa sylvatica</i> Marsh.	99	518	119	301	39	68
	<i>Prunus serotina</i> Ehrh.	124	49	221	122	114	61
		Total	394	714	513	622	200
<i>Pinus</i> L.	<i>Pinus taeda</i> L.	28	21	65	54	86	90
	<i>Pinus palustris</i> Mill.	4	0	12	13	13	8
		Total	32	21	77	68	99

Table A-3 continued.

Species group	Species	1991		1994		2002	
		Dormant	Growing	Dormant	Growing	Dormant	Growing
Understory woody dicots (UWD)	<i>Aralia spinosa</i> L.	0	0	1	0	0	0
	<i>Baccharis halimifolia</i> L.	0	0	0	0	0	1
	<i>Ilex vomitoria</i> Ait.	0	0	0	0	9	0
	<i>Ligustrum</i> L.	0	0	0	5	3	0
	<i>Myrica cerifera</i> L.	0	0	0	0	4	11
	<i>Prunus virginiana</i> L.	0	0	0	0	4	2
	<i>Viburnum</i> L.	0	0	0	0	40	17
	<i>Celtis</i> L.	2	2	0	0	1	0
	<i>Tilia americana</i> var. <i>caroliniana</i> (Miller) Castigl.	15	2	77	0	0	0
	<i>Ilex opaca</i> Ait.	25	12	41	23	31	19
	<i>Crataegus</i> L.	54	35	112	55	78	26
	<i>Sassafras albidum</i> (Nutt.) Nees	92	124	12	33	3	7
	<i>Rhus</i> L.	260	797	338	1047	48	199
	<i>Cornus</i> L.	306	5	368	4	105	1
<i>Vaccinium arboreum</i> Marsh.	1016	1296	1425	1369	608	1083	
Total	1770	2272	2373	2536	934	1364	
Unknowns	Unknowns	0	0	1	7	4	5
Grand Totals		3415	3823	4087	4046	1710	727

Table A-4. Mean number of stems per hectare by level of burn severity.

Species category	Species	1991			1994			2002		
		High	Low	No	High	Low	No	High	Low	No
<i>Quercus</i> L.	<i>Quercus alba</i> L.	111	157	154	8	15	3	0	2	4
	<i>Quercus coccinea</i> Muenchh.	24	10	33	15	4	4	1	0	0
	<i>Quercus falcata</i> Michx.	196	294	255	89	270	130	26	60	55
	<i>Quercus laevis</i> Walt.	0	0	0	1	0	1	61	60	24
	<i>Quercus marilandica</i> Muenchh.	53	11	15	60	26	12	23	16	23
	<i>Quercus nigra</i> L.	3	6	23	21	41	96	44	53	51
	<i>Quercus stellata</i> Wangenh.	72	99	85	112	174	163	10	22	9
	<i>Quercus velutina</i> Lam.	26	26	53	43	17	47	10	1	5
	Total	484	602	619	349	547	456	175	215	170
<i>Carya</i> Nutt.	<i>Carya</i> Nutt.	506	508	363	420	731	431	195	257	195
Other potential overstory species (OPO)	<i>Diospyros virginiana</i> L.	61	35	44	25	28	42	16	6	9
	<i>Fraxinus</i> L.	0	0	3	0	0	0	0	0	0
	<i>Liquidambar styraciflua</i> L.	62	127	104	69	252	116	31	49	12
	<i>Nyssa sylvatica</i> Marsh.	373	257	295	237	182	210	59	49	51
	<i>Prunus serotina</i> Ehrh.	59	95	105	117	221	177	74	97	96
	<i>Ulmus alata</i> Michx.	23	3	14	18	3	6	14	3	5
	Total	578	517	566	467	686	550	193	205	173
<i>Pinus</i> L.	<i>Pinus taeda</i> L.	0	0	6	3	0	35	9	1	20
	<i>Pinus palustris</i> Mill.	21	20	32	76	57	46	87	100	78
	Total	21	20	38	80	57	81	97	101	98

Table A-4 continued.

Species category	Species	1991			1994			2002		
		High	Low	No	High	Low	No	High	Low	No
Understory woody dicots (UWD)	<i>Aralia spinosa</i> L.	0	0	0	0	1	0	0	0	0
	<i>Baccharis halimifolia</i> L.	0	0	0	0	0	0	1	0	0
	<i>Celtis</i> L.	3	2	0	0	0	0	2	0	0
	<i>Cornus</i> L.	196	128	142	202	180	176	77	41	41
	<i>Crataegus</i> L.	12	50	71	56	110	85	29	63	63
	<i>Ilex opaca</i> Ait.	5	32	20	9	62	24	20	29	25
	<i>Ilex vomitoria</i> Ait.	0	0	0	0	0	0	0	11	3
	<i>Ligustrum</i> L.	0	0	0	7	0	0	0	5	0
	<i>Myrica cerifera</i> L.	0	0	0	0	0	0	14	5	2
	<i>Prunus virginiana</i> L.	0	0	0	0	0	0	1	6	2
	<i>Rhus</i> L.	636	462	487	801	674	602	129	145	96
	<i>Sassafras albidum</i> (Nutt.) Nees	40	119	164	15	14	40	1	3	11
	<i>Tilia americana</i> var. <i>caroliniana</i> (Miller) Castigl.	1	1	24	0	1	115	0	0	0
	<i>Vaccinium arboreum</i> Marsh.	1431	964	1072	1228	1299	1663	1087	700	749
	<i>Viburnum</i> L.	0	0	0	0	0	0	40	40	4
Total	2325	1760	1979	2318	2340	2705	1401	1050	996	
Unknowns	Unknowns	0	0	0	1	0	11	1	7	6
	Grand Totals	3914	3407	3565	3634	4361	4234	2062	1835	1638

Table A-5. Mean number of stems per hectare by pine regeneration treatment.

Species group	Species	1991		1994		2002	
		Natural	Planted	Natural	Planted	Natural	Planted
<i>Quercus</i> L.	<i>Quercus laevis</i> Walt.	0	0	2	0	43	53
	<i>Quercus nigra</i> L.	13	8	52	53	38	60
	<i>Quercus marilandica</i> Muenchh.	22	30	42	24	24	17
	<i>Quercus coccinea</i> Muenchh.	24	21	12	3	1	0
	<i>Quercus velutina</i> Lam.	40	30	41	30	8	3
	<i>Quercus stellata</i> Wangenh.	64	106	126	173	10	18
	<i>Quercus alba</i> L.	121	160	5	12	3	1
	<i>Quercus falcata</i> Michx.	225	272	136	190	40	54
	Total	510	627	416	485	167	205
<i>Carya</i> Nutt.	<i>Carya</i> Nutt.	390	529	483	573	177	250
Other potential overstory species (OPO)	<i>Fraxinus</i> L.	1	1	0	0	0	0
	<i>Liquidambar styraciflua</i> L.	24	2	14	3	12	3
	<i>Diospyros virginiana</i> L.	56	37	22	42	10	11
	<i>Nyssa sylvatica</i> Marsh.	87	86	176	168	87	88
	<i>Prunus serotina</i> Ehrh.	109	87	206	86	39	23
	<i>Ulmus alata</i> Michx.	372	244	261	159	70	38
	Total	650	458	678	458	218	163
<i>Pinus</i> L.	<i>Pinus palustris</i> Mill.	4	0	20	6	18	3
	<i>Pinus taeda</i> L.	12	37	62	58	109	70
	Total	16	37	82	63	127	73

Table A-5 continued.

Species group	Species	1991		1994		2002	
		Natural	Planted	Natural	Planted	Natural	Planted
Understory woody dicots (UWD)	<i>Aralia spinosa</i> L.	0	0	1	0	0	0
	<i>Baccharis halimifolia</i> L.	0	0	0	0	0	1
	<i>Ilex vomitoria</i> Ait.	0	0	0	0	5	4
	<i>Ligustrum</i> L.	0	0	5	0	3	0
	<i>Myrica cerifera</i> L.	0	0	0	0	3	11
	<i>Prunus virginiana</i> L.	0	0	0	0	3	3
	<i>Viburnum</i> L.	0	0	0	0	7	48
	<i>Tilia americana</i> var. <i>caroliniana</i> (Miller) Castigl.	2	15	1	77	0	0
	<i>Celtis</i> L.	3	1	0	0	0	1
	<i>Ilex opaca</i> Ait.	20	17	39	24	31	19
	<i>Crataegus</i> L.	43	46	83	84	48	55
	<i>Sassafras albidum</i> (Nutt.) Nees	74	142	28	17	1	8
	<i>Cornus</i> L.	243	69	269	103	77	31
	<i>Rhus</i> L.	547	510	721	664	126	121
	<i>Vaccinium arboreum</i> Marsh.	1210	1101	1341	1452	607	1059
Total	2142	1901	2487	2422	911	1361	
Unknowns	Unknowns	0	0	8	0	9	1
	Grand Totals	3708	3552	4154	4001	1609	2053

Table A-6. ANOVA results suggesting that sites (replicates) strongly influenced group densities. An * indicates significant results ($p \leq 0.05$), ns = not significant.

Year		<i>Quercus</i>		<i>Carya</i>		OPO		<i>Pinus</i>		UWD	
		F	Pr > F	F	Pr > F	F	Pr > F	F	Pr > F	F	Pr > F
1991	Season*Replicate	6.33	0.008 *	ns	ns	ns	ns	ns	ns	ns	ns
1994	Replicate	ns	ns	18.5	0.000 *	0.260	0.772	23.35	0.000 *	4.33	0.029 *
	Season*Replicate	5.42	0.014 *	ns	ns	ns	ns	7.55	0.004 *	ns	ns
	Burn*Replicate	ns	ns	ns	ns	ns	ns	3.09	0.042 *	ns	ns
2002	Replicate	ns	ns	ns	ns	ns	ns	4.04	0.036 *	ns	ns

Table A-7. ANOVA results suggesting that sites (replicates) strongly influenced diversity. An * indicates significant results ($p \leq 0.05$), ns = not significant.

Year		Richness		Shannon index		Equitability	
		F	Pr > F	F	Pr > F	F	Pr > F
1991	Replicate	6.73	0.001 *	5.44	0.014 *	5.44	0.014 *
1994	Replicate	4.09	0.034 *	5.56	0.013 *	5.56	0.013 *
	Season*Burn*Replicate	ns	ns	2.97	0.048 *	2.97	0.048 *

Table A-8. Full ANOVA results listed by species group using relative density data for each group. An * indicates significant results ($p \leq 0.05$).

Year		<i>Quercus</i>		<i>Carya</i>		OPO		<i>Pinus</i>		UWD	
		F	Pr > F	F	Pr > F	F	Pr > F	F	Pr > F	F	Pr > F
1991	Season	4.59	0.046 *	5.70	0.028 *	4.72	0.043 *	1.48	0.239	3.08	0.010 *
	Burn	0.95	0.404	1.36	0.282	0.07	0.937	1.77	0.198	1.44	0.260
	Replicate	0.11	0.675	2.41	0.119	1.87	0.183	0.12	0.890	4.83	0.690
	Season*Burn	0.40	0.893	1.36	0.282	0.04	0.961	0.75	0.489	0.38	0.020 *
	Season*Rep	6.33	0.008 *	0.54	0.593	0.54	0.589	0.96	0.402	1.87	0.180
	Burn*Rep	0.30	0.874	0.43	0.783	1.23	0.332	0.36	0.835	2.13	0.120
	Season*Burn*Rep	1.99	0.140	1.24	0.329	1.04	0.412	0.20	0.935	0.97	0.450
	Pine regeneration	1.21	0.280	2.72	0.11	1.70	0.202	7.12	0.012 *	0.70	0.410
1994	Season	3.51	0.077	1.88	0.187	0.36	0.555	0.41	0.532	0.04	0.851
	Burn	1.71	0.210	4.98	0.019 *	0.50	0.615	1.13	0.344	0.40	0.675
	Replicate	0.19	0.826	18.5	0.000 *	0.26	0.772	23.35	0.00 *	4.33	0.029 *
	Season*Burn	0.73	0.496	1.40	0.271	0.18	0.836	0.67	0.523	0.96	0.401
	Season*Rep	5.42	0.014 *	1.02	0.379	1.00	0.389	7.55	0.004 *	0.70	0.508
	Burn*Rep	0.17	0.952	2.34	0.094	1.12	0.378	3.09	0.042 *	1.17	0.359
	Season*Burn*Rep	1.35	0.292	0.96	0.453	0.25	0.903	0.98	0.445	0.85	0.510
	Pine regeneration	0.45	0.510	0.62	0.436	1.78	0.193	0.93	0.342	0.00	0.950
2002	Season	7.31	0.015 *	2.37	0.141	7.31	0.015 *	0.01	0.932	1.26	0.276
	Burn	0.77	0.477	0.46	0.639	0.77	0.477	0.02	0.983	0.44	0.653
	Replicate	0.37	0.695	1.41	0.271	0.37	0.695	4.04	0.036 *	2.67	0.097
	Season*Burn	0.84	0.447	0.02	0.978	0.84	0.447	1.78	0.197	0.09	0.914
	Burn*Rep	1.59	0.220	0.38	0.819	1.59	0.220	1.16	0.359	0.62	0.656
	Season*Burn*Rep	1.72	0.190	0.41	0.801	1.72	0.190	0.14	0.965	0.63	0.645
	Pine regeneration	1.06	0.311	1.79	0.191	1.06	0.311	11.69	0.002 *	1.99	0.168

Table A-9. Full ANOVA results of diversity indices: richness, Shannon-Wiener index (H'), and equitability (J). An * indicates significant results ($p \leq 0.05$).

Year	Treatment	Richness		Shannon index		Equitability	
		F	Pr >F	F	Pr >F	F	Pr >F
1991	Season	9.82	0.006 *	5.38	0.032 *	5.38	0.032 *
	Burn	4.55	0.025 *	4.72	0.022 *	4.72	0.022 *
	Replicate	6.73	0.001 *	5.44	0.014 *	5.44	0.014 *
	Season*Burn	0.30	0.748	0.83	0.453	0.83	0.453
	Season*Rep	0.48	0.628	0.52	0.603	0.52	0.603
	Burn*Rep	0.85	0.513	0.42	0.794	0.42	0.794
	Season*Burn*Rep	0.64	0.643	0.61	0.663	0.61	0.663
	Pine regeneration	0.68	0.419	0.00	0.946	0.00	0.947
1994	Season	4.24	0.054 *	1.29	0.271	1.29	0.271
	Burn	2.04	0.159	0.82	0.457	0.82	0.457
	Replicate	4.09	0.034 *	5.56	0.013 *	5.56	0.013 *
	Season*Burn	0.15	0.858	0.74	0.491	0.74	0.491
	Season*Rep	0.35	0.707	0.94	0.410	0.94	0.410
	Burn*Rep	1.29	0.312	0.79	0.546	0.79	0.546
	Season*Burn*Rep	1.37	0.284	2.97	0.048 *	2.97	0.048
	Pine regeneration	1.00	0.331	0.35	0.561	0.35	0.560
2002	Season	9.96	0.007 *	5.20	0.038 *	5.20	0.038 *
	Burn	0.20	0.821	0.02	0.976	0.02	0.976
	Replicate	2.96	0.082	0.90	0.428	0.90	0.428
	Season*Burn	1.62	0.230	0.07	0.936	0.07	0.936
	Season*Rep	0.23	0.795	0.54	0.592	0.54	0.592
	Burn*Rep	0.43	0.782	0.43	0.788	0.43	0.788
	Season*Burn*Rep	0.75	0.574	1.09	0.395	1.09	0.395
	Pine regeneration	0.31	0.867	0.42	0.529	0.42	0.529

VITA

Kimberly Crider was born in Crown Point, Indiana on January 4, 1975. She attended public schools in Crown Point, Indiana and graduated in 1993 from Crown Point High School. That same year she entered Indiana University, in Bloomington, Indiana. She received a Bachelor of Science in Environmental Science in May, 1997. During this time she attained work experience with the National Park Service and the Nature Conservancy. In August 2000, she entered the Master's program in geography at the University of Tennessee. During this time she served as a graduate teaching assistant to introductory geography courses. The Master of Science degree was received in May, 2003. In the spring of 2001 she was hired by the USDA Forest Service, Southern Research Station in Athens, Georgia and presently works there as a research assistant.

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