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I am submitting herewith a thesis written by Samuel Jackson entitled "First-year Changes in Oak Regeneration, Understory Competitors, and Resource Levels in Response to Two Overstory Treatments and Prescribed Burning at Chuck Swan State Forest." 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 Forestry.

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**FIRST-YEAR CHANGES IN OAK REGENERATION, UNDERSTORY
COMPETITORS, AND RESOURCE LEVELS IN RESPONSE TO TWO
OVERSTORY TREATMENTS AND PRESCRIBED BURNING AT CHUCK
SWAN STATE FOREST**

**A Thesis
Presented for the
Master of Science
Degree
The University of Tennessee, Knoxville**

**Samuel Wayne Jackson
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DEDICATION

This thesis is dedicated to my wife and family. Daphne, you have believed in me, supported, encouraged, and inspired me to do my best and to never give up. I love you with all my heart. To my parents, Wayne and Julie Jackson, I thank you for supporting me in whatever I chose to do and for your encouragement in reaching my goals. To my entire family, including my in-laws, I thank you for supporting Daphne and me through so much over the last few years. We would not have been able to accomplish what we have without your support and encouragement.

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ABSTRACT

Oaks (*Quercus spp.*) are highly valuable as sources of forest products, in promoting recreation, and in providing food and habitat for wildlife. Oak regeneration has significantly declined over the past century in many regions of the eastern United States. This has led to increased concern with respect to the maintenance of oak-dominated ecosystems on the landscape in the future. It is currently thought that oaks are being out competed by species that either grow faster or are more tolerant of shade than oaks. Many of these competitors are more sensitive to fire than oak, and it is thought that human exclusion of fire from the landscape has resulted in an increase in the abundance of these competitors. Prescribed burning, herbicides, and cutting are potential methods of reducing the abundance of competitors, but these methods have not been definitively tested in all regions of the eastern U.S., and have produced mixed results in some cases.

I investigated the effects of four treatments on forest vegetation, particularly impacts on oak species and their primary competitors. These treatments were: shelterwood cutting, wildlife thinning using herbicide, wildlife thinning using herbicide in conjunction with prescribed burning, and prescribed burning with no overstory treatment. Herbs, woody regeneration, and woody stems in the midstory and overstory were all monitored before and after treatment to quantify changes in plant species composition, cover, density, and structure. Post-treatment light and soil moisture were also measured to help explain vegetation responses.

Shelterwood harvests and wildlife thinnings increased understory light availability while showing no statistically significant effects on soil moisture levels. Both treatments significantly reduced overstory cover as well as midstory vegetation. Prescribed fire was

found to significantly increase the density of oak seedlings and sprouts < 10-cm tall. Red maple regeneration, the most abundant competitor prior to treatment, was reduced in all size classes by prescribed burning. Two other primary competitors, sassafras and yellow-poplar, were both significantly increased in all regeneration size classes.

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I. INTRODUCTION AND REVIEW OF RESEARCH

The Importance of Oaks

Eighty-nine percent of Tennessee's 14.1 million acres of forestland is classified as hardwood forest with forty-five percent of the hardwood timber volume being comprised of oaks (Schweitzer, 2000). Nationally, oak ecosystems occupy 114 million acres, and provide habitat for more than 75 associated tree species and 230 species of wildlife (Rogers et al, 1993). However great the impact of humans on oaks, this genus is one of the most dominant in our forests today.

Oaks form a cornerstone of Tennessee's economy. The forest industry provides almost 70,000 jobs with an output of 9.1 billion dollars (Idassi et al, 1998). In 1997, oaks accounted for 48.5% of the state's total hardwood sawtimber output, a volume of 77,222,000 cubic feet (Stratton and Wright, 1999). Using the average price for oak sawtimber in 1997, \$242 per thousand board feet, that output was worth an estimated \$112,126,344 to Tennessee's economy (Hodges, unpublished data). Oaks accounted for 50.2% of the volume of the total hardwood output for the year 1997, both sawtimber and pulpwood (Stratton and Wright, 1999).

Oak forests provide a number of economic benefits and values beyond forest products. Forests are used by society for a multitude of different purposes, including bird watching, hunting, hiking, and camping. Oaks also enhance the aesthetic qualities of forests and provide food and shelter for wildlife. The total economic impact oaks have on society is difficult to estimate, but exceeds that of the economic impact of oak timber alone.

Since the decline of the American chestnut and the subsequent loss of chestnut mast, wildlife have become more dependent on oak mast in the southeast. Many use it as the primary source of food in their diet. Studies on rumen samples have found that white-tailed deer (*Odocoileus virginianus*) use acorns for 50 to 100 percent of their diet and bobwhite quail (*Colinus virginiana*) have been shown to use acorns for up to 46 percent of their diet (Goodrum et al, 1971). Other wildlife such as wild turkey (*Meleagris gallopavo*) and squirrels (*Sciurus spp.*) have also been shown to depend on oaks for significant parts of their diets (Dickson, 2001; Goodrum et al, 1971). Van Dersal (1940), through a review of Biological Survey files, found 186 different species of wildlife feed on various tissues of oak.

Historical Development of Southeastern Oak Forests

Oak (*Quercus* spp.) forests have been a dominant forest type in eastern North America for centuries (Delcourt and Delcourt, 1997). As a group, angiosperms appeared in the Southeast during the Mesozoic Era and most species now present in the region were present at the end of the Pleistocene era (Buckner, 1989). After the advance and retreat of the glaciers 10,000 years ago, plant distributions closely resembled what they are today (Buckner, 1989). Humans also began to affect southeastern forests at this time. Dickens (1976) stated that humans were found in the southern Appalachians as early as 12,000 years ago. Studies have shown that oaks were a dominant species by at least 10,000 years ago (Watts, 1979). Human effects on disturbance regimes in our forests became apparent at this time (Buckner, 1989). Delcourt et al (1986) reported that eastern Tennessee was predominantly oak and pine at least 1500 years ago and that evidence of fires increased with increasing human populations. Keel (1976) also noted a substantial

increase in the occurrence of fire at this time. Fire was a tool that Native Americans used for hunting and clearing vegetation (DeVivo, 1990, Van Lear and Waldrop, 1989). Research on the pre-European settlement forests has shown that oaks continued to be a dominant species during the period of Native American fire use (Russell, 1980). As the region that would become the eastern United States was settled in the 18th and 19th centuries, many areas experienced an increase in the dominance of oak due to a variety of human disturbances. Land clearing, grazing, and burning favored oaks by reducing competitors. However, by 1920, attitudes had changed, and the United States Forest Service was opposed to fire in the forests (Pyne, 1982). Fire exclusion, disease introduction, and other human impacts have altered the distribution and densities of oak over the last two centuries, both positively and negatively (Abrams, 1986; Brose et al, 2001; Rogers et al, 1993). Today, there are estimated to be over 500 species of oak worldwide (Rogers and Johnson, 1998). Oaks are considered to be the most widely dispersed hardwood genus in the northern temperate zone and in North America alone there are an estimated 200-250 species of oaks (Larsen and Johnson, 1998; Van Dersal, 1940).

Effects of a Growing Nation

Settlers are thought to have affected oaks, primarily oak regeneration, by exclusion of fire during the late 1800s and most of the 1900s. As civilization spread on the North American continent, an important issue was controlling disturbances, such as fire and flooding, so as to limit hazards to humans. Wildfire suppression quickly became a standard practice as fires were quickly put out wherever they were found. Abrams (1986) stated that after Europeans colonized the United States, wildfire decreased due to

road building, town expansion, intensive agriculture, and active fire suppression. This suppression now appears to have significant effects on oaks in terms of competition control and regeneration recruitment into the overstory (Abrams, 2000, Brose et al., 2001).

Another prominent human effect on forests with a major oak component was the introduction of an exotic fungus. The introduction of the chestnut blight (*Endothia parasitica*) in the early part of the twentieth century contributed to the decline of American chestnut (*Castanea dentata*) and a corresponding increase in the dominance of oaks (Abrams, 2000). Oaks gradually replaced American chestnut, and are currently a vital source of forest products and food for wildlife in the eastern forests.

The Current Oak Regeneration Problem

Oaks are not maintaining and regenerating themselves in southeastern forests. They are being replaced by later successional species (Abrams, 2000). This problem has become more evident since the 1970s (Lorimer, 1989). Due to the importance of oaks, both ecologically and economically, it is widely held that the failure of oaks to regenerate is the major problem facing silviculture today (Van Lear, 1990). Oaks are generally favored by disturbance, such as periodic fire (Abrams, 1992). Much of the oak dominated forest we have today developed following human disturbances such as clearing land for agriculture, burning, and lumbering (Clark, 1993, Crow, 1988). Abrams (2000) showed that northern red oak expanded greatly in northern hardwood forests following European settlement. Oaks took advantage of the increased light and growing space created by these disturbances and grew into the open canopies. However, some of the very disturbances that favored oaks also spurred the conservation movement of the 1930s and

1940s (Clark, 1993; Rogers and Johnson, 1998). Extensive land clearing and logging left cutover lands in ruin with problems such as erosion, loss of organic matter, and poor water quality. Over the last 70 years, conservation efforts have reduced disturbances in both frequency and scale. Lorimer (1992) revealed, through a review of literature, that the oak regeneration problem has developed and become serious during the same time period. As previously discussed, oaks are generally favored by disturbances including land clearing, fires, and grazing. At the turn of the twentieth century, researchers began to notice a lack of oak regeneration on the forest floor, much of which was due to excessive fires and grazing (Lorimer, 1989). In 1907, Greely and Ashe stated that the primary key to successfully establishing oak regeneration was protecting stands from fire. Oak regeneration developed, yet many of its competitors developed at much faster and efficient rates. By the 1950s and 60s, researchers began finding that oaks comprised a small percentage of advanced regeneration in the understory and were failing to regenerate in existing oak stands (Weitzman and Trimble, 1957, Gammon et al, 1960, McGee and Hooper, 1970). Lorimer (1989) states that there is “no longer any question that oak will be displaced on many sites.” As the oak stands created by frequent burning and other disturbances matured and developed dense canopies, later successional shade tolerant and fire-intolerant species, such as maple (*Acer spp.*) and beech (*Fagus spp.*) began to dominate understory regeneration (Abrams, 1992). The lack of major canopy disturbance has begun to allow these species to replace oaks in the overstory (Crow, 1988). When a stand is harvested, the oak component in the new stand often declines, relative to the original stand (Smith, 1993). This lack of regeneration has been attributed to natural successional processes resulting from a lack of major disturbances on the site

(Lorimer, 1993; Rogers et al, 1993). Oak is generally considered a mid-successional species, and without disturbance the oak-dominated forests will continue to progress into forests dominated by species such as sugar maple (*Acer sacharrum*) and American beech (*Fagus grandifolia*), both of which are late successional and more shade tolerant than oaks. The primary body of evidence covering oak regeneration problems suggests that problems exist more on high quality sites where certain competitors, e.g. yellow-poplar (*Liriodendron tulipifera*) in openings and red maple (*Acer rubrum*) in closed stands, have a competitive advantage over oaks (Oak, 1993). Oak regeneration is often more successful on poor quality sites, where large root systems and other adaptations to drought give oak the advantage over competitors. Weitzman and Trimble (1957) found that oaks successfully regenerate when site index is 65 and lower.

Human influence on natural disturbance regimes seems to have been a primary cause of this regeneration problem. The fire suppression of the 1800s and 1900s has removed a key disturbance that helped perpetuate the mid-successional state beneficial to oaks. Crow (1988) stated that the “abundance of oaks throughout the Eastern Deciduous Forest is an artifact of disturbance regimes that are no longer common (primarily due to fire suppression).” Oaks are adapted to fire by means of thick bark, large root systems, and a root collar that is located beneath the ground, all of which lead to vigorous sprouting after top kill by a fire (Kolb and Steiner, 1990; Lorimer, 1985; Van Lear and Watt, 1992). Due to large root systems and ample stored reserves, oaks can sprout back more vigorously after fire than competitors that allocate a greater proportion of resources to shoots. (Crow, 1988, Kolb and Steiner, 1990). Lorimer (1985) showed that fire in low frequencies may have been an important factor in forming current mature oak stands by

reducing competition and favoring oaks. Some of the primary competitors of oak, such as red maple (*Acer rubrum*) and yellow-poplar (*Liriodendron tulipifera*) have thin bark, small root systems, and root collar buds above ground. Fires are more likely to kill and reduce the densities of these competitors. Reductions in fire-intolerant competitors should be accompanied by increases in space, light, moisture, and nutrient availability. Studies in the northeastern U.S. have demonstrated that stands once dominated by sugar maple were converted to northern red oak (*Quercus rubra*) stands by frequent fires (Kline, 1985). In a study area without fire disturbance, 70% of the oak regeneration died after a clearcut due to competition from other vegetation (Beck and Hooper, 1986). The more tolerant and more competitive species that replace oaks are not desired because they are not as valuable, both in terms of economic or wildlife values (Lorimer, 1989). Tests of prescribed fire have shown that the proportion of oak regeneration in a stand can be increased without new seedling establishment as the competing vegetation is killed while oaks sprout back (Brose and Van Lear, 1998; Swan, 1970).

The result of most disturbances, whether fire or another event, is to increase light availability, reduce competition, and increase the growing space of oaks. Even though oaks are intermediate in tolerance of shade, prolonged shading can lead to seedling mortality (Crow, 1988). Carvell and Tryon (1961) indicated that of all the environmental factors at a particular site, light intensity is most highly correlated with the amount of oak seedlings. Their study also showed that oak generation was more abundant on disturbed sites than undisturbed sites. However, Crow (1988) analyzed the results of several studies and concluded that the levels of light are not the limiting factor of oak seedling establishment. Other factors, such as soil moisture, nutrient levels, and site conditions,

have a more significant impact on seedling establishment than light. Tubbs (1977) showed that after a seedling establishes itself with a good root system, light then becomes a more limiting factor that impedes growth. Crow (1992) made this clear by showing light intensity was positively correlated with the amount of oak seedling growth. Once seedlings expend the energy reserves of the root system, lack of light can result in mortality. Beck (1981) found that survival rates of red oak regeneration were very low in natural stands, with only 38% surviving after six growing seasons. He reported that 60% of the mortality occurs in the first two growing seasons after establishment. For those seedlings that do survive in the understory for a longer period of time, many are unable to respond quickly enough to outgrow their competitors when released by competition control (Crow, 1988). Beck (1970) found that simply removing the low, competing vegetation was not enough. Seedlings only respond to low competition control when part of the overstory is also removed. On a site without competition control, he reported that competing vegetation had over-topped red oak seedlings by the fifth growing season. Light intensity can also control competition between oaks and certain competitors. Kolb et al. (1990) found that shade reduced yellow-poplar growth more than northern red oak growth, thus giving the oak some growth advantage in partial shade. In full light, yellow-poplar outgrew the oaks in the study. Although light is important, it is common to find some oaks, such as chestnut oak (*Quercus montana*), white oak (*Quercus alba*), and northern red oak (*Quercus rubra*) doing best under partial shade. Red oak, one of the oaks more intermediate in shade tolerance, has been shown to be more stressed for moisture and nutrients in fully open areas than in partially shaded areas (Kolb et al., 1990).

Other problems, in addition to lack of disturbance, can also lead to poor oak regeneration locally. Seed production in oaks is incredibly variable (Lorimer, 1989), with bumper seed crops occurring every 2 to 10 years (Sander et al 1983). The lack of seed could result in poor local oak regeneration. Wildlife regularly predate acorns and can reduce oak germination on a site even in years of good acorn production. Rodents such as squirrels and chipmunks use acorns as a primary part of their diets, as do turkeys and deer. White-tailed deer also browse oak seedlings (Beck and Della-Bianca, 1981). Deer use acorns as a primary food source in the fall, but during late winter and early spring, when the acorn supply is limited, they will use woody browse as a food source. Marquis (1981) found that in areas where deer were excluded, oak regeneration stems over 5 feet tall were three times as plentiful as in areas where deer could browse. This deer browsing reduced the stocking of both preferred wildlife and commercial species on the site. That same study also showed that the estimated mature commercial value of the stand browsed by deer was half of the value of the unbrowsed stand, at maturity. Rodents can also kill newly established seedlings by digging their root systems up to feed on the acorn still attached to the root system (Beck and Della-Bianca, 1981). A number of insect species can destroy the cotyledons in an acorn. Two of these insects, the acorn weevil (*Curculio spp.*) and the moth *Melissopus latiferreanus*, are considered to be the most damaging insects (Marquis et al, 1976). These insects lay their eggs inside the acorn and the larvae will eat the cotyledons and embryo as they develop.

Local environmental conditions can also play a small role in oak regeneration. Red oaks have shown decreased root growth when subjected to period of drought

conditions (Larson and Whitmore, 1970). However, other studies have shown oak species, such as white oak, to be insensitive to moisture conditions (Ferrell, 1953).

Possible Solutions to the Problem

The dominant forests in the southeast today are comprised primarily of species that depend upon disturbance for regeneration (Buckner, 1989). If disturbances, such as fire, are not introduced or allowed to naturally return to these forests, the composition could change drastically over the next century. Not only will management strategies need to include disturbances, but they will also have to address other factors that affect regeneration such as insects and wildlife in order to successfully regenerate oaks.

Several studies have dealt with examining ways to increase the amount and competitive ability of oak regeneration. Research has shown that ensuring and enhancing the development of adequate advanced oak regeneration is the best way to ensure oak regeneration after a disturbance. Protecting and aiding the development of that regeneration is an important goal of forest managers. Larsen and Johnson (1998) state that the regeneration of oak forests depends upon advanced regeneration accumulated over several successful acorn crops and in good site conditions. These conditions include adequate nutrient supply, adequate growing space, and low densities of competitors. Acorns are not always the sole source of regeneration. Advanced regeneration can develop from sprouts that originate from dormant buds in the vicinity of the root collar after browsing, cutting, or other disturbances. During the harvest of oak stands, advanced regeneration is often damaged, leading to resprouting (Sander, 1971). Sander (1972) also showed that advanced regeneration needed to be of a large size to be successfully recruited into the overstory. If the regeneration is too small, harvesting should be delayed

to allow growth to a sufficient size (Sander, 1972). The ability of oaks to die back and resprout enable them to survive for long periods in the understory, even though many oak species are intermediate in tolerance of shade (Rogers et al., 1993). Oak seedling survival is related to site characteristics such as overstory and understory densities, light, moisture, frost, and predation by animals. Managing oak regeneration, therefore, depends on addressing these site characteristics to make those sites more favorable to oaks in order to successfully recruit advanced regeneration into the overstory.

Since oaks have been shown to respond favorably to natural disturbances, it is important for regeneration treatments to mimic those disturbance types and regimes. Most disturbances create openings in the forest canopy, ranging from very small gaps to total removal of the canopy, which allow increased light to reach the forest floor. Light intensity is one of the easiest forest characteristics to manipulate (Phares, 1971). Today's forest management mimics natural disturbances through harvesting mature trees and thus increasing light intensity on the forest floor. Single-tree selection, group selection, shelterwood cuts, and clearcuts are all methods of partially or totally removing the canopy. Single-tree selection creates small canopy gaps as individual trees are removed and group selection creates larger gaps by removing groups of several trees in a confined area. A shelterwood cut generally removes a percentage of the canopy evenly across a given area, while clearcuts remove the entire canopy. Research has shown that single-tree selection does not create canopy openings large enough to facilitate successful oak regeneration, and favors more shade-tolerant species as a result (Larsen and Johnson, 1998). A similar problem can occur with group selection. Hill and Dickmann (1988) concluded that group selection cuts produce less reproduction of red oaks than

shelterwood cuts or clearcuts. Clearcutting has been shown to be an effective way to regenerate oaks, usually in conjunction with some type of control of yellow-poplar and other competitors (Larsen and Johnson, 1998). Clearcuts, however, are not aesthetically pleasing to the public and have been shown to expose oak seedlings to frost damage in the Lake States (Buckley et al., 1998). Clearcutting is also less cost effective than other cuts (Loftis, 1990). Crunkilton et al. (1992) found that water relations within individual red oaks were better in clearcuts than in shelterwood cuts. This stress also caused poor root development in shelterwood cuts. This was attributed to the fact that more moisture is generally available in a clearcut than a shelterwood cut (Buckley et al., 1998; Lee, 1980). However, shelterwoods do exhibit some increase in soil moisture over uncut stands (Buckley et al., 1998). Annand and Thompson (1997) found that tree regeneration heights were greater in shelterwood cuts and clearcuts than in the single-tree selection and group selection treatments. Maximum energy production for red oaks has been documented at around one-third partial light, a level achieved only by a harvest such as a shelterwood cut (Phares, 1971). For these and other reasons, shelterwood harvests have been recommended for regenerating oaks and other hardwoods by several researchers (Brose and Van Lear, 1998; Buckley et al., 1998; Johnson et al., 1998; Kellison et al., 1981; Loftis, 1990).

A shelterwood harvest is defined as a regeneration cut that is conducted to establish a new, even-aged community beneath the partial canopy of residual trees, which are removed later to release the regeneration in the understory and midstory (Nyland, 1996). Brose et al. (1999a) performed a shelterwood cut as a harvest technique where 40-50% of the basal area of a stand was removed. They recommend that the stand be

relatively well-stocked, have basal area of at least 23 m² per hectare, and have substantial advanced oak regeneration. Also, trees left should be of high quality, vigor, and of the desired species, presumably oaks. Shelterwoods allow existing oak regeneration to further develop by disturbing the soil and increasing light availability to the forest floor (Hill and Dickmann, 1988). Johnson et al. (1998) found that regeneration 2-4 years after a shelterwood cut was very similar to that of a clearcut; a mix of intolerant and tolerant species. After the initial harvest, the remaining overstory can be left for as little as 5 years or as long as 13 years (Loftis, 1990). This allows the advanced oak regeneration to develop an advantage over competitors such as yellow-poplar, which does not grow well in partial light. Loftis (1990) stated that shelterwood cuts “provide stand conditions that will enhance the development of established red oak advanced reproduction.” Yet most studies state that even with the success of the shelterwood cut, some type of competition control is still needed to give oaks a competitive advantage (Pubanz et al, 1989; Ray et al, 1999). Crow (1992) found that a combination of partial overstory removal and competition control in the understory increased growth and reduced mortality of oak seedlings.

Controlling competing species can be done in several ways. Oaks compete with a variety of species in different regions. In northern forests of the eastern United States, oaks tend to compete with American beech, sugar maple, and red maple (Lorimer, 1984; Ray et al, 1999). In southern forests, oaks tend to compete with red maple as well as yellow-poplar and other fast growing species (Kays et al, 1988; O'hara, 1986). Many types of competition control have been studied in both regions with similar results, regardless of the region. Herbicides have been used to reduce the density of the

understory to allow oaks more light. Yet, Kass and Boyette (1998) found that herbicides did not work due to poor kill rates after application. Also, herbicides are very expensive and can become cost prohibitive when applied to large areas. The most effective methods of competition control are those that selectively impact certain species or groups of species due to the adaptations of those species. Prescribed fire is a prime example of this and a very effective way of controlling oak competition.

Fire as a Regeneration Tool

Oaks are considered to be adapted to fire. Studies have been conducted to determine appropriate timing and burn intensity for facilitating oak regeneration (Barnes and Van Lear, 1998; Huddle and Pallardy, 1999). Barnes and Van Lear (1998) found that a single burn conducted during early spring is equally as effective at reducing midstory/understory density and competition as three successive winter burns. All fires, no matter which season, were low intensity, with flame heights averaging about 30.5-cm. The fires of this study moved at a relatively rapid pace of 1.52 to 3.05 meters per minute. Huddle and Pallardy (1999) also found that oaks resprouted better after spring fires than autumn fires. In terms of controlling competing vegetation, this study showed that although burning reduced survival in all species, oak survival was 2 to 10 times that of red maple. However, yellow-poplar and red maple have been found to resprout significantly after one fire. Several successive burns can help significantly reduce resprouting in these species (Barnes and Van Lear, 1998). Apart from reducing competitors of oak, fire also affects other aspects of forest vegetation and characteristics such as herbaceous plants, soil moisture, and light availability. Much of that impact can depend on the maximum temperature a fire reaches. Franklin et al (1997) found that

maximum fire temperature decreased as height above ground increased. Temperatures on that study ranged from 52 to 260 degrees Celsius. Generally, the hotter the fire, the more top killing and direct effects are seen.

Burning can affect soil properties. Boerner (2000) concluded that although soil temperatures increase after burning, these increases are not enough to cause heat mortality to plants already present and/or re-colonizing the site. Prescribed burning can increase soil pH and increase microorganisms within the soil (Barnes et al., 1998). They concluded that overall, fire does not have a detrimental effect on soils and their processes. Boerner et al (1988) reported that soil nutrient levels did not significantly change with burning. Phillips et al (2000) found that 35 years of burning at 5-year intervals had “little long-term effects on the surface horizons” of the soil. However, they did conclude that prescribed fire reduced the organic and A-horizon layers. Burning has also been shown to reduce soil moisture levels in some cases (Barnes and Van Lear, 1998). Burning could increase soil moisture due to killing plants, but at the same time could reduce it due to evaporation.

Burning can also have a significant impact on the herbaceous and understory vegetation of a forest. Fires reduce the density of small diameter stems in the midstory. Work by Arthur et al. (1998) demonstrated that sites that were not burned had a midstory density 2 to 6 times greater than burned sites. DeSelm et al (1973) found grass and forb coverage increased with an increased frequency in burning, and annual burning is conducted to maintain and restore prairie ecosystems. DeSelm and Clebsch (1991) found that herbaceous plant cover increased each year after a fire while cover decreased on control plots. Elliot et al (1999) found that species richness and diversity were

significantly lower on sites in western North Carolina after burning, but cover of deciduous shrubs, such as blueberry, and grasses increased. The density of red maple seedlings increased after fires along with oaks. Over time, however, red maple had a higher mortality rate than all other species (Elliot et al., 1999). Fires significantly affect understory regeneration in Rhode Island (Brown, 1960). On burned areas, primary species in the regeneration layer included scarlet oak (*Quercus coccinea*), while areas that were not burned contained greater amounts of red maple and white oak. The study found that typical sprouters in a burned area included scarlet, black (*Quercus velutina*), and white oaks (*Quercus alba*). Brown (1960) concluded that areas where burns occurred will remain predominantly oak.

A concern of prescribed fires is that they can also damage existing overstory trees by causing wounds and introducing rot into the boles, thus making them less valuable for commercial markets. The extent of fire wounding depends on several factors including weather, fire behavior, and bark characteristics (Sutherland and Smith, 2000). Most prescribed fires are not intense enough to cause extensive wounding, although wounding can occur with flare-ups and hot spots. During prescribed burning after a shelterwood cut, 85 percent of residual oaks were either slightly damaged or not damaged at all (Brose and Van Lear, 1999). Oak trees that are damaged in prescribed fire seem to be able to readily compartmentalize wounded tissues and protect the remaining bole tissues of the tree. Abrams (1996) found that white oaks were fairly resilient to wounding by rapidly compartmentalizing the wound to prevent decay of the stem. In general, fires do not cause significant bole damage to residual trees, provided the fire intensity is kept low and fire-adapted species are present and of sufficient age.

The Shelterwood-Burn Technique

Shelterwood methods have been successful when conducted in association with a type of competition control. Several recent papers have been published describing a shelterwood-burn technique in detail (Brose and Van Lear, 1998; Brose and Van Lear, 1999; Brose et al., 1999a; Brose et al., 1999b). The majority of the research was conducted on productive upland oak sites where competition among oaks and other species is most intense. The stands in the study area all had basal areas greater than 22.96 m² per hectare, were at least 80 years old, and had high site index values (Brose et al., 1999a). Also, all stands had pre-existing oak regeneration in the understory. The shelterwood harvest removed 40-60 percent of the basal area and the trees left in the overstory were species such as oak. Trees removed included yellow-poplar and other undesirables. Slash was left on site with care not to pile debris against residual trees, which would allow intense fire to damage the tree. Brose et al. (1999a) recommend waiting 3 to 5 years after the partial harvest before conducting the prescribed burn. This delay allows existing regeneration, both oak and undesirable species, to respond to the increased light availability. Beck (1970) determined that the best time for release from competing vegetation is four to five years after the harvest. Waiting any longer than that could be detrimental to the oak seedlings. Fuel also builds up in the form of leaf litter during this time. The recommended season to burn is spring, for maximum kill of competitors (Brose and Van Lear, 1998). Brose et al. (1999a) recommend a slow moving fire with flame heights not over 1.22-m. Brose et al. (1999b) found that oak stocking exceeded yellow-poplar stocking in the burned areas while the reverse happened in the control areas. Red maple has also been shown to be susceptible to prescribed fire (Brose

et al., 1999a). It has been noted that more than one subsequent burn may be necessary to sufficiently control competition in order for oak regeneration to have the maximum advantage.

Some very positive effects have been shown by fire and shelterwood cuts. Keyser et al. (1996) showed a reduction of up to 96% in oak competitors such as red maple and yellow poplar. Oaks decreased by a much smaller proportion, 11%. Oak and hickory (*Carya spp.*) regeneration was shown to exhibit accelerated growth for at least the first two seasons after a burn while competitors only showed increased growth for one growing season at most (Brose and Van Lear, 1998). The benefit of this method in regenerating oaks is well documented by the studies referenced above. The shelterwood cut increases light to the forest floor while the fire controls oak competition in the understory.

Implications of Regeneration Treatments

As with any kind or type of forest management, factors other than the trees themselves are affected by regeneration treatments. Wildlife, for example, is directly affected by all types of forest management. Several studies have been conducted to analyze wildlife responses to various forest management techniques (Annand and Thompson, 1997; Artman et al., 2001; Brennan et al., 1998; Huntley and McGee, 1981; Johnson et al., 1995; King and DeGraaf, 2000). Two years after harvest, shelterwood cuts had higher bird species richness than clearcuts, single-tree selection, and group selection treatments in southeastern Missouri (Annand and Thompson, 1997). King and DeGraaf (2000) found similar patterns in northern New Hampshire and also found that clearcuts and mature forests did not differ in species richness. They found that shelterwood cuts

showed the greatest potential for obtaining high bird species richness. This richness was attributed to the fact that shelterwood cuts provide habitat for birds that prefer mature forest habitat as well as those that prefer the early successional, young forests. Predation of nests did not change with increased forest management, yet patterns suggested that the increased understory development of shelterwood stands might actually decrease nest predation. Shelterwoods and clearcuts can also provide the needed canopy openings to encourage early successional forbs and browse development, which are both important in white-tailed deer and other wildlife diets.

Prescribed burning can also greatly affect wildlife habitat. Short-term results of prescribed burning on breeding bird populations showed no negative effects (Artman et al., 2001). However, that same study speculated that if fire was implemented on a frequent basis (yearly), some populations of ground nesting birds could be reduced. No studies on longer term fire frequencies (5 years and up) exist. Brennan et al. (1998) noted that several species have undergone significant declines in population levels due to the fire suppression of the last century. They attribute declines in northern bobwhite, red-cockaded woodpecker, Bachman's sparrow, and other wildlife populations to the absence of fire. These and many other species depend upon the open, early successional habitat created by fires. Prescribed fire has also been shown to provide increased browse for forest herbivores and to increase herbaceous cover plants beneficial to wildlife (Huntley and McGee, 1981).

Another factor that has to be considered when conducting forest management is public policy concerning management activities. Prescribed burning is regulated as to when and where burning can occur. Laws exist in most states requiring burning permits

and regulating the time of year burning can occur, so as to minimize the risk of burning during the high fire risk seasons. There are also serious social concerns over fire management. Studies have shown that the public has several concerns with fire. Weldon (1996) reported that the public was concerned with the effects of smoke on health and visual quality as well as the idea that fire “wasted” wood fiber by needlessly destroying it. Social concerns also vary by region. More than 70 percent of the prescribed burning in the United States occurs within the southeast (Ward et al., 1993). For example, a study of southern Alabama private, non-industrial, forest landowners showed that over 70 percent of them saw prescribed burning as an important forest management activity on their land (McConnell and Baldwin, 1991). That same study indicated that over 60 percent of people who had neighbors that used prescribed burning for management felt “positively influenced” by the neighbor’s use of burning. In some areas, though, prescribed fire may not be an option. Areas close to development such as roads and homes are less likely to be burned due to concerns of fire escape and smoke spread that can lead to health and visibility problems. In areas like these, treatments other than prescribed burning need to be used.

One alternative method of overstory removal and midstory competition control is the use of herbicides in forest management. By girdling and injecting trees with herbicides, overstory density can be greatly reduced, thus allowing increased light to the forest floor. However, herbicides do have a variable rate of kill on trees to which they are applied, unlike a timber harvest where tree removal is guaranteed (Kass and Boyette, 1998). This differential effectiveness makes herbicides a less reliable method of canopy thinning. Herbicide use is less labor intensive than harvesting, but is much more costly as

it generates no revenue. However, herbicides can increase the quality of wildlife habitat. By increasing light to the forest floor, using herbicides to kill overstory and mid-story stems can increase the abundance of understory herbaceous plants. McComb and Rumsey (1981) concluded that herbicide application to trees while selecting to remove red maple, yellow poplar, and pines (*Pinus* spp.) may improve mast production and foliage growth of oaks. Trees that are treated and left to stand after their death provide important habitat for insects and feeding/habitat areas for many birds. Besides the benefits of herbicides, there are some concerns with the side effects of application. Many times herbicides are applied in such small amounts that the chemicals will not affect vegetation other than the target trees, and thus will not affect wildlife or water quality. Herbicides, when applied according to the recommended procedures from the manufacturer, generally do not have significant effects on the environment and efficiently increased light and growing space for vegetation (Morrison and Meslow, 1983).

Research Needs

Research to date has investigated the use of prescribed fire, various types of timber harvesting, and herbicides in several states in the eastern United States as tools for enhancing oak regeneration (Barnes and Van Lear, 1998; Boerner et al., 1988; Brose et al., 1999a; Elliot et al., 1999; Kass and Boyette, 1998; Swan, 1970; Van Lear, 1990). Other research has assessed the use of these management tools for improving wildlife habitat by encouraging the development of understory vegetation and structure (Annand and Thompson, 1997; Artman et al., 2001; Brennan et al., 1998; Goodrum et al., 1971; Johnson et al., 1995; King and DeGraaf, 2000; Morrison and Meslow, 1983).

Prescribed burning research has returned mixed results due to differences in the timing and intensity of fires, site characteristics such as site quality, and pre-burn species composition. Spring burns have generally produced the best results in terms of encouraging oaks by controlling competition (Brose and Van Lear, 1998). Summer, fall, and winter burns are more difficult due to seasonal weather patterns and do not reduce oak competition as well as spring burns (Brose and Van Lear, 1999a). Fire intensity, in terms of both flame height and fire temperature is highly variable from site to site. Fuel types and loadings, topography, and weather can all influence fire intensity on a given site. Fire intensity differs regionally, by timber type, and by season. Site quality affects the presence and composition of advanced regeneration, something that must be present on a site prior to burning. High quality sites often support a greater density of competitors of oak than low quality sites (Weitzman and Trimble, 1957). Research results from similar burning treatments can vary based on site quality. Pre-burn species composition can also influence results. Sites that have little to no oak regeneration prior to treatment may not respond as well as sites that have adequate advanced oak regeneration. If sites are heavily composed of oak competitors, response could be similar. Oaks have to be present on the site in order for burning to give them competitive advantage over their competitors.

The effects of harvesting and/or herbicide use tend to vary due to regional differences in factors that can affect oak regeneration. Buckley et al. (1998) found that deer browsing can play a significant role in reducing success of oak regeneration, depending upon the level of disturbance and the overstory composition. That study also revealed that frost damage could harm regeneration with increasing overstory removal. In

high quality forest stands where oak competitors, such as yellow poplar, are present before cutting, overstory removal can stimulate the growth of those competitors (Loftis, 1990).

Without a definite and clear solution to the current oak regeneration problem, there is a need for continued testing of current methods in different regions, especially promising methods such as the shelterwood-burn technique. This testing is needed to explore the effects of various factors, such as species composition, deer abundance, climate, and site characteristics, that differ between regions, on the effectiveness of various techniques used for enhancing oak regeneration and wildlife habitat. For example, Brose et al. (1999a) show the benefits of the shelterwood-burn technique in South Carolina. Some factors that were not considered in that study include the following. If deer densities are high in an area managed with this technique, does the deer browse negate any potential benefits by killing the oak sprouts that develop? Does this technique work equally well on different sites in different regions of the eastern United States? Also, the role pre-treatment species composition plays needs further study. Does this composition affect the overall vegetative response to the treatment? These and many other questions currently remain unanswered.

In 2000, the opportunity arose to examine the effects of several potentially interacting factors in a study of the effectiveness of prescribed burning, shelterwood cutting, and herbicide treatment of selected overstory and mid-story stems in enhancing wildlife habitat.

II. RESEARCH OBJECTIVES AND GOALS

The objectives in this research were two-fold. One objective was to record the changes in forest vegetation, primarily regeneration of oaks and their competitors, that occurred due to the treatments of shelterwood cutting, wildlife thinning with herbicides, and prescribed burning. This included documenting overstory structure, mid-story structure, and herbaceous plants. The second objective was to record changes in light availability and soil moisture levels in response to the treatments. Specific objectives of this research were 1) to compare and evaluate the effectiveness of shelterwood harvests and wildlife thinnings with herbicides in stimulating advanced oak regeneration; 2) to test the hypothesis that prescribed fire enhances the positive effects of these treatments on oak regeneration; 3) to establish and document the impact of all treatment combinations on herb, shrub, and tree species that are potential competitors of oak regeneration; and 4) to document light and soil moisture availability associated with the treatment combinations to further explain treatment effects on oak regeneration and other forest vegetation.

III. MATERIALS AND METHODS

Study Area

This study was conducted at Chuck Swan State Forest and Wildlife Management Area in eastern Tennessee, near the town of Sharp's Chapel. It is located in Claiborne and Union counties and the forest comprises approximately 24,000 acres of a peninsula in Norris Lake. The Tennessee Valley Authority (TVA) purchased the tract in 1934 as part of the land acquisition for the construction of Norris Dam. Initially named the Central Peninsula Forest, the area was managed by the Tennessee Department of Conservation. Around the time of acquisition, experimental forestry studies and inventories were implemented. Wildlife management and recreation development began in 1947 and in 1952, the state of Tennessee purchased the area from TVA. It employed a Forest Ranger to manage the area and in 1973, an on-site forester was placed on the forest as a full-time manager. Today, the area is co-managed by the Tennessee Division of Forestry and the Tennessee Wildlife Resources Agency with both agencies having full-time managers on site, as well as a crew of technicians. The area is managed for timber production and recreation such as hunting, hiking, and wildlife viewing.

The primary timber type of Chuck Swan is mixed hardwoods. Predominant species include white oak, chestnut oak (*Quercus montana*), black oak, scarlet oak, blackgum (*Nyssa sylvatica*), red maple, and yellow-poplar. There is also a large component of planted pine on the area consisting mainly of loblolly pine (*Pinus taeda*) and eastern white pine (*Pinus strobus*). The terrain of Chuck Swan is considered to be hilly with elevations ranging from 305 meters above sea level to over 488 meters above sea level. Temperatures on the area range from a yearly average high of 20.4 degrees

Celsius to a yearly average low of 7.9 degrees Celsius. The area receives approximately 1193.8-mm of rain per year (NCDC, 2001).

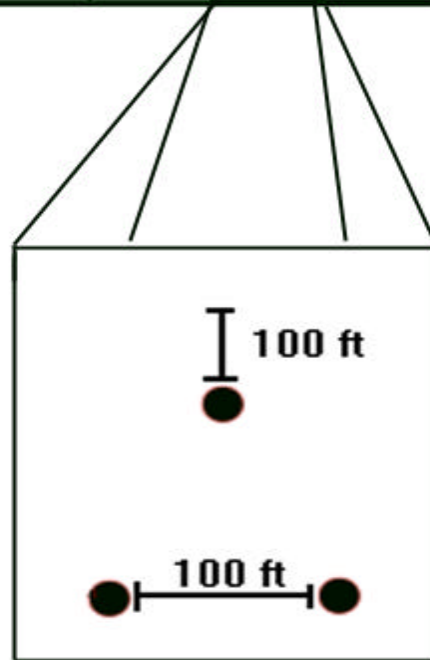
Four 9.7-hectare stands on moderately productive sites were delineated for study. All study sites were located on northwest aspects, with slopes averaging 24 percent to 30 percent. Stand 1 was located on the northwest side of the peninsula while stand 2 was located on the southeastern side of the forest. Stand 3 was located on the northeastern side of the forest and stand 4 was located on the southwestern side of the forest. All stands were comprised of mixed (oak-hickory) hardwoods with a small component of pine.

Treatments

Five different treatment combinations and a control were used in this study. The treatments were a prescribed burn alone with no overstory treatment, shelterwood harvest unburned, shelterwood harvest with prescribed burning, wildlife thinning with herbicides unburned, and wildlife thinning with herbicides combined with prescribed burning. Each treatment was randomly assigned to a 0.81-hectare cell. Each treatment was assigned to two cells in each stand (Figure 1).

The wildlife thinnings with herbicide were the first treatments conducted. These thinnings were designed to enhance habitat and food production for wildlife. No economic or timber production concerns were considered. Trees that were marked to be removed were trees with lesser benefit to wildlife, such as red maple and yellow-poplar. A target residual basal area of 11.5 square meters of basal area per hectare was established. In some cases, on good sites, oaks had to be removed to achieve the target basal area. The herbicide used on this research was Garlon 3A, manufactured and

			SB	SNB	UCB
			C	WB	WNB



C = Control

UCB = Uncut Prescribed Burned

SB = Shelterwood Cut Designated to be Burned

SNB = Shelterwood Cut Designated to Remain Unburned

WB = Wildlife Thinning Prescribed Burned

WNB = Wildlife Thinning Unburned

Figure 1. Treatment and plot design along with treatment abbreviation key.

provided by Dupont Chemicals. The herbicide was applied in a 50/50 mixture with water. Herbicide was applied in two primary ways. On trees that had relatively large diameters, a chainsaw was used to completely girdle the tree and herbicide was then applied to the cut. For trees that were smaller in diameter, a chainsaw, hatchet, or brush blade were used to cut the tree down. The stump was then treated with the herbicide to prevent resprouting. No trees or debris were removed from the site. All stands had herbicide applied in late February - March 2001.

The prescribed burning was conducted on the uncut treatments and the wildlife thinning treatments in April 2001. Stand 2 was burned on April 9. The day's high temperature was 28.9 degrees Celsius and the low was 13.3 degrees. There was no precipitation within 3 days of the burn. Stand 1 was burned on April 10 along with one-half of stand 3. The high temperature that day was 28.9 degrees Celsius with a low of 13.3 degrees as well. There was no precipitation for 4 days prior to the burn. The second half of stand 3 was burned on April 20 with a high temperature of 17.8 degrees Celsius and a low of 3.9 degrees. No measurable precipitation had occurred for at least 3 days prior to burning. Stand 4 was burned on April 23 and 27. April 23 had a recorded high temperature of 28.3 degrees Celsius and a low of 12.2 degrees. April 27 had a high temperature of 22.2 and a low of 8.3 degrees Celsius. Neither burning day had any precipitation for at least six days before the burn. Measurements for temperature and precipitation were recorded at the Maynardville, TN weather station. Measurements of relative humidity were recorded on site at the burns as well as at the McGhee-Tyson Airport weather station in Knoxville, TN. Fire intensity was estimated based on average flame heights. Flame heights for most burned areas averaged 3-4 feet from the ground. In

stand 3, however, an uncut burned treatment cell that had been affected by a windstorm a decade ago did not carry a consistent fire. The treatment cell was dominated by saplings and ferns, which seemed to hold more moisture than the other areas burned.

The shelterwood harvests were conducted in early summer 2001. Timber to be cut was marked in May 2001. This marking was done based on timber production and regeneration goals. The goal of the harvest was to have a residual basal area of approximately 11.5 square meters per hectare. Trees that were left were of good quality, as vigorous as possible, and of good form. Whenever possible, oak stems were left as residual stems. Trees removed included American beech, red maple, sassafras (*Sassafras albidum*), and hickory. The harvesting was done by B.J. Fortner Hardwoods Inc., a contract logging and lumber company. The harvesting crew included one sawyer in the woods, one person operating a bulldozer for skidding, and one person located on the landing for log loading and trimming. Logging was done from June 19 until July 20. To be consistent with the shelterwood-burn recommendations proposed by Brose et al. (1999a), the shelterwoods will be prescribed burned in 3 to 5 years.

Measurements

Pre-treatment data were collected in August, September, and early October 2000. Treatments were installed during early 2001 and similar post-treatment data were collected in August and September 2001.

Within each treatment cell, 3 permanent sampling plots were established. To prevent edge effect and measurement overlap, each plot was at least 30.5 meters from the cell edge and at least 30.5 meters from another plot. Figure 2 represents the measurement plot design.

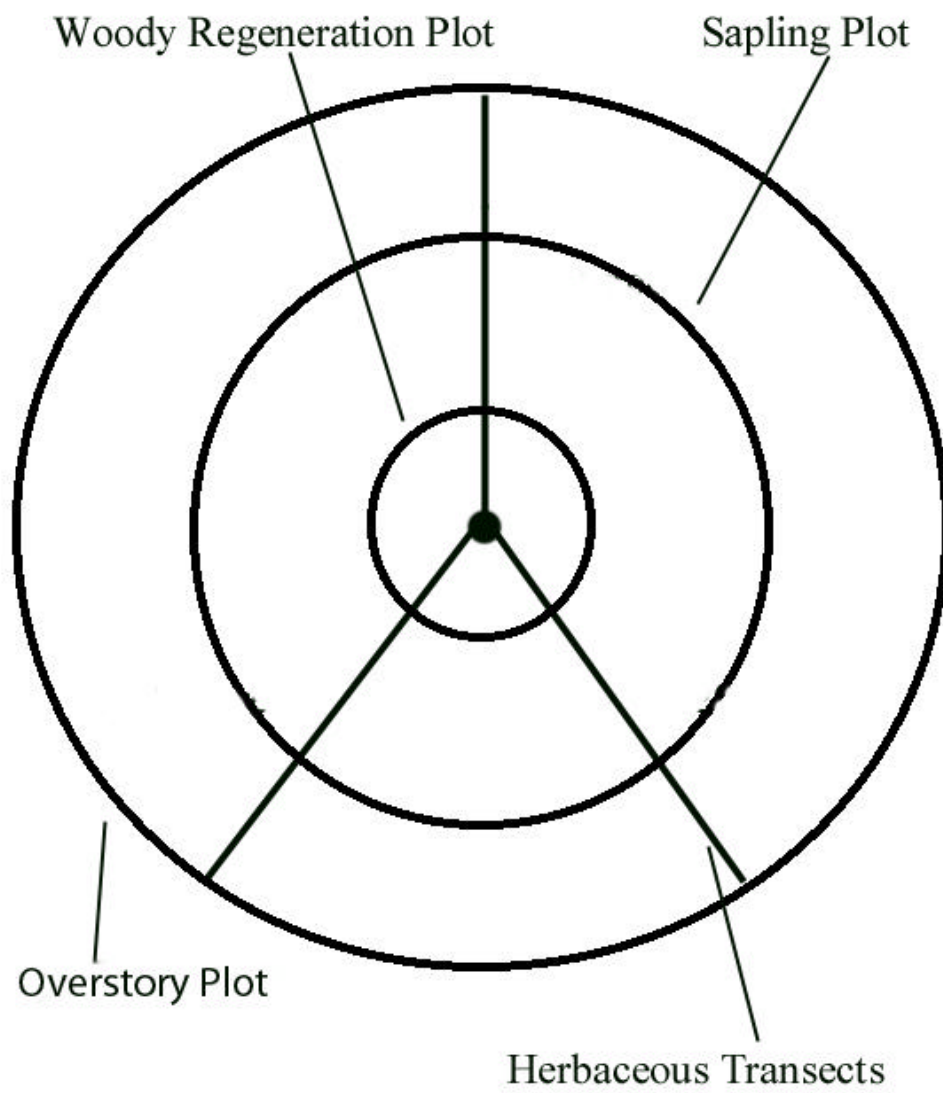


Figure 2. Measurement plot design.

Herbaceous plants were measured in terms of species and coverage at each sampling plot. Three transects, 11.3 meters long, were established around plot center on bearings of 0, 120, and 240 degrees. A metric tape was stretched the length of the transect. Wherever an herbaceous plant was under the tape, its species and coverage of the tape in centimeters were recorded.

Woody regeneration was measured by using a 3.6-meter radius circular plot. Within this plot, all woody stems less than 1.4 meters tall were counted. The count was organized by species and then by height class. Two height classes were used for woody regeneration: stems less than 10-cm tall, and those between 10-cm and 1.4-m tall.

Woody saplings were measured within a 5.7-m radius circular plot. All woody stems greater than 1.4-m tall and less than 11.4-cm in diameter at breast height (DBH) were measured. The data was organized by species.

The percent of the forest floor covered by the crowns of shrubs less than 5 meters tall was visually estimated in the 5.7-m radius plot. The percent of the shrub crown cover comprised of soft-mast producers was also visually estimated. Basal area was measured using a 10 basal area factor prism from plot center. Slope was measured using a handheld clinometer (Suunto Corp.) at each plot.

Handheld spherical densiometer (Forest Densiometers Inc.) readings were used to estimate canopy cover. Readings were taken at points 5.6 meters from plot center in each of the four cardinal directions. When stationed at each direction, four measurements were taken, one in each cardinal direction, for a total of 16 measurements per plot.

Overstory tree data was recorded within an 11.3-m radius circular plot. Tree diameter and species were recorded for trees larger than 11.4-cm in DBH. Basal area of the study sites was also calculated using the diameters of the overstory trees.

Post-treatment measurements of photosynthetically active radiation (PAR), the wavelength of light that plants use to conduct photosynthesis, were taken with an AccuPAR Ceptometer (Decagon Devices Inc.; Pullman, WA). Measurements were taken in September 2001 along the same three transects used for herbaceous measurements. Measurements were taken at plot center and at the mid-point and end-point of each transect. Two measurements were taken at each point. One measurement was taken at 32.5-cm above the ground, to measure light at the regeneration level. The other measurement was taken at approximately 1-m above ground to measure light at the shrub level. Reference measurements were taken in the open just before and after sampling in the stands to calculate percent full PAR at each measurement point. Light measurements were taken from 10:00 am until 2:00 pm to minimize effects of sun angle.

Post-treatment soil moisture measurements were collected over 2 days in October 2001. No measurable precipitation had occurred within 4 days of the measurements. Percent volumetric soil moisture was measured using a Trase TDR soil moisture probe (Soil Moisture Equipment Corp; Goleta, CA). One measurement was taken at each plot center, for a total of three per treatment cell.

Data Analysis

Data were averaged by plot and analyzed using Analysis of Variance (General Linear Model (GLM) procedure, SAS Institute, 2000). Class variables used were stand and treatment. The General Linear Model tested for differences within years between

stands and treatments using the stand-treatment interaction as an error term. Means were separated using Tukey's mean separation technique. The alpha level of significance used was $P < 0.05$. The Shapiro-wilk test, the W value, was used to test the normality of the data. A W of 0.90 or higher indicated normally distributed data. Most of the results presented here represent the data by years. Year 2000 is the pre-treatment year and year 2001 is the post treatment year. To conserve space, both years will be presented in the text and only year 2001 results will be presented graphically. Year 2000 results will be presented in the text as a baseline.

Both the shelterwood cut to be burned and the shelterwood cut to remain unburned were expected to have similar results in each analysis, since the prescribed burning had not been conducted, in accordance with the Brose et al. (1999a) prescription of waiting 3-5 years after the initial harvest to burn the site.

IV. RESULTS

Overstory Tree Data Analysis

To analyze the overstory data, several dominant species groups were selected for analysis. Two major characteristics were analyzed. The first was the amount of basal area per hectare of that species within a particular treatment. The other was the percent of the total basal area for the treatment that the specific species made up. Table 1 lists the species of overstory trees recorded on the study area.

For pre-treatment American beech, no significant treatment differences ($P=0.7159$) were detected in basal area of the species. Treatment means ranged from 1.14 to 3.18 square meters of American beech basal area per hectare. Similar results were found for pre-treatment American beech as a percent of total basal area, no significant differences were found between treatments ($P=0.4327$). Treatment means ranged from 5 to 17 percent of total basal area. American beech basal area for pre-treatment year 2001 did not significantly differ between treatments ($P=0.5923$). Treatment means ranged from 1.24 to 3.97 square meters of American beech basal area per hectare. Post-treatment American beech as a percent of total basal area for year 2001 showed no significant differences between treatments ($P=0.5121$). Treatment means ranged from 7 to 20 percent of the total basal area.

For pre-treatment blackgum basal area, no significant differences between treatments ($P=0.1391$) were detected. Treatment means ranged from 0.98 to 2.44 square meters of blackgum basal area per hectare. Similar results were found for pre-treatment blackgum as a percent of total basal area. No significant treatment differences ($P=0.5121$)

Table 1. Overstory tree species list.

Scientific Name	Common Name	Scientific Name	Common Name
<i>Acer rubrum</i>	Red Maple	<i>Nyssa sylvatica</i>	Blackgum
<i>Acer saccharum</i>	Sugar Maple	<i>Oxydendrum arboreum</i>	Sourwood
<i>Carpinus caroliniana</i>	American Hornbeam	<i>Pinus echinata</i>	Shortleaf Pine
<i>Carya glabra</i>	Pignut Hickory	<i>Pinus virginiana</i>	Virginia Pine
<i>Carya ovata</i>	Shagbark Hickory	<i>Prunus serotina</i>	Black Cherry
<i>Carya tomentosa</i>	Mockernut Hickory	<i>Quercus alba</i>	White Oak
<i>Cornus florida</i>	Flowering Dogwood	<i>Quercus coccinea</i>	Scarlett Oak
<i>Diospyros virginiana</i>	Common Persimmon	<i>Quercus falcata</i>	Southern Red Oak
<i>Fagus grandifolia</i>	American Beech	<i>Quercus montana</i>	Chestnut Oak
<i>Fraxinus americana</i>	White Ash	<i>Quercus rubra</i>	Northern Red Oak
<i>Juglans nigra</i>	Black Walnut	<i>Quercus stellata</i>	Post Oak
<i>Juniperus virginiana</i>	Eastern Red Cedar	<i>Quercus velutina</i>	Black Oak
<i>Liriodendron tulipifera</i>	Yellow-Poplar	<i>Sassafras albidum</i>	Sassafras
<i>Magnolia acuminata</i>	Cucumber Magnolia	<i>Tilia americana</i>	American Basswood
<i>Morus rubra</i>	Red Mulberry		

were found. Treatment means ranged from 3 to 10 percent of the total basal area. No significant treatment differences were found in either blackgum basal area ($P=0.0647$) or blackgum as a percent of total basal area ($P=0.3878$) for post-treatment year 2001. For blackgum basal area, treatment means ranged from 0.97 to 2.61 square meters per hectare. For blackgum as a percent of total basal area, treatment means ranged from 4 to 11 percent.

No significant treatment differences were detected in black oak basal area in either year 2000 ($P=0.6022$) or year 2001 ($P=0.8667$). Black oak basal area treatment means ranged from 4.38 to 8.47 square meters per hectare in year 2000 while year 2001 basal area averaged 4.04 to 7.49 square meters. No significant treatment differences were detected in black oak as a percent of the total basal area in either year 2000 ($P=0.3759$) or year 2001 ($P=0.9390$). The black oak as a percent of total basal area treatment means ran from 15 to 33 percent while means for year 2001 ranged from 18 to 27 percent of the total basal area.

No significant treatment differences ($P=0.5869$) were detected in pre-treatment chestnut oak basal area. Treatment means ranged from 6.29 to 11.75 square meters of chestnut oak basal area per hectare. No pre-treatment significant differences between treatments ($P=0.7235$) were detected for chestnut oak as a percent of total basal area as treatment means ranged from 22 to 36 percent of the total basal area. Post-treatment chestnut oak basal area did not differ significantly between treatments ($P=0.8810$). Treatment means ranged from 7.76 to 10.70 square meters of chestnut oak per hectare. In post-treatment chestnut oak as a percent of total basal area, treatments were not

significantly different ($P=0.8385$). The chestnut oak as a percent of total basal area treatment means ranged from 32 to 45 percent.

For mockernut hickory basal area, no significant differences between treatments were found in either year 2000 ($P=0.5831$) or 2001 ($P=0.6592$). Treatment means ranged from 0 to 3.09 square meters per hectare of pre-treatment mockernut hickory basal area while post-treatment basal area ranged from 0.41 to 2.19 square meters per hectare. For mockernut hickory as a percent of total basal area, no significant differences were found in year 2000 ($P=0.5466$) or 2001 ($P=0.3101$). Pre-treatment means ranged from 0 to 12 percent and post-treatment means ranged from 2 to 16 percent of the total basal area made up of mockernut hickory.

For northern red oak basal area, no significant differences between treatments were found in either year 2000 ($P=0.4908$) or 2001 ($P=0.4571$). Treatment means ranged from 2.33 to 6.52 square meters per hectare of pre-treatment northern red oak basal area while post-treatment basal area ranged from 1.10 to 4.82 square meters per hectare. For northern red oak as a percent of total basal area, no significant differences were found in year 2000 ($P=0.7721$) or 2001 ($P=0.4845$). Pre-treatment means ranged from 10 to 20 percent and post-treatment means ranged from 6 to 19 percent of the total basal area made up of northern red oak.

No significant treatment differences were detected in pignut hickory basal area in either year 2000 ($P=0.8263$) or year 2001 ($P=0.6130$). Pignut hickory basal area treatment means ranged from 1.18 to 2.76 square meters per hectare in year 2000 while year 2001 basal area averaged 1.66 to 3.99 square meters per hectare. No significant treatment differences were detected in pignut hickory as a percent of the total basal area

in either year 2000 ($P=0.7280$) or year 2001 ($P=0.5901$). The pignut hickory as a percent of total basal area treatment means ran from 5 to 13 percent while means for year 2001 ranged from 6 to 20 percent of the total basal area.

No significant treatment differences were detected in pine species group basal area in either year 2000 ($P=0.6171$) or year 2001 ($P=0.1952$). Pine species group basal area treatment means ranged from 0.65 to 3.37 square meters per hectare in year 2000 while year 2001 basal area averaged 0 to 2.65 square meters per hectare. No significant treatment differences were detected in pine species group as a percent of the total basal area in either year 2000 ($P=0.6303$) or year 2001 ($P=0.1566$). The pine species group as a percent of total basal area treatment means ran from 3 to 14 percent while means for year 2001 ranged from 0 to 16 percent of the total basal area.

There were no pre-treatment significant differences between treatments in red maple basal area ($P=0.0836$) or red maple as a percent of total basal area ($P=0.1392$). Treatment means for red maple basal area ranged from 2.55 to 5.27 square meters per hectare and the means for red maple as a percent of total basal area ran from 11 to 24 percent. In post-treatment red maple basal area, significant differences between treatments (.0338) were found (Figure 3). The unburned wildlife thinning (5.31 square meters/ha) had significantly higher red maple basal area than the shelterwood cut designated to be unburned (1.32 square meters/ha). Post-treatment red maple as a percent of total basal area exhibited no significant differences between treatments. Treatment means ranged from 8 to 31 percent of the total basal area made up of red maple.

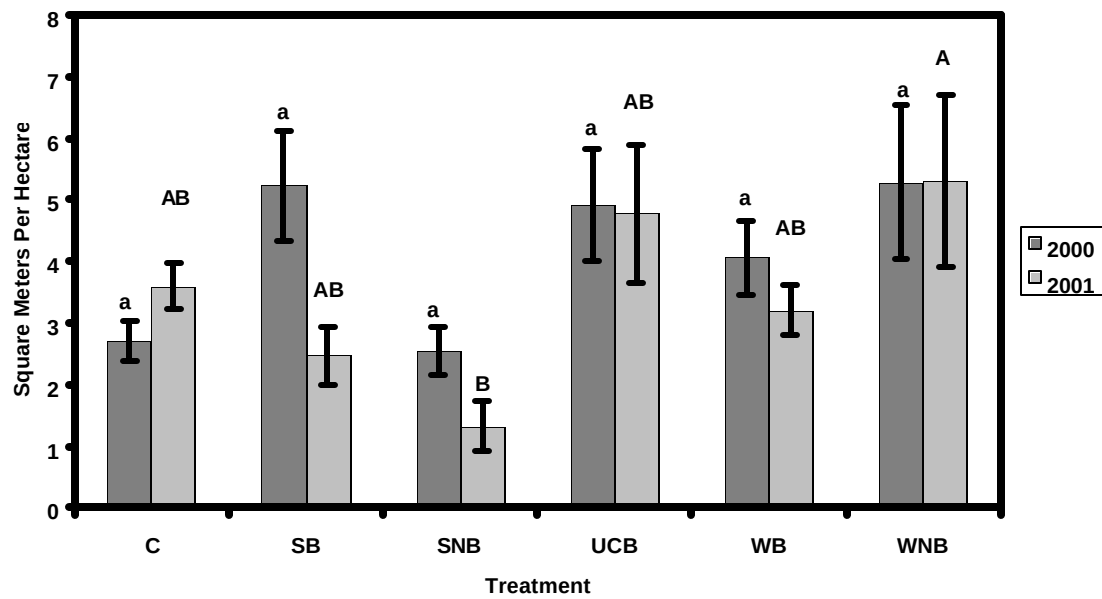


Figure 3. Red maple basal area by treatment. Error bars represent one standard error beyond the mean. Bars with similar letters are not significantly different within years. Lower case letters were used with year 2000 and capital letters were used with year 2001 data. If no letters are present for a particular set of bars, no statistically significant separation was found.

For sassafras basal area, no significant differences between treatments were found in either year 2000 ($P=0.2700$) or 2001 ($P=0.8129$). Treatment means ranged from 0.28 to 1.54 square meters per hectare of pre-treatment sassafras basal area while post-treatment basal area ranged from 0.27 to 2.02 square meters per hectare. For sassafras as a percent of total basal area, no significant differences were found in year 2000 ($P=0.2956$) or 2001 ($P=0.5755$). Pre-treatment means ranged from 3 to 10 percent and post-treatment means ranged from 1 to 9 percent of the total basal area made up of sassafras.

For pre-treatment scarlet oak basal area, treatment differences were significant ($P=0.0080$). The shelterwood treatment designated to be burned (8.04 square meters/ha) had significantly more basal area than the shelterwood cut designated to remain unburned (2.17 square meters/ha) and the uncut burned areas (1.88 square meters/ha). The prescribed burned wildlife thinning (7.64 square meters/ha) had significantly more basal area than the uncut burned treatment (1.88 square meters/ha). For pre-treatment scarlet oak as a percent of total basal area, treatment differences were significant ($P=0.0044$). The scarlet oak in the prescribed burned wildlife thinning (36 %) made up a significantly higher percent of total basal area than the uncut burned area (9 %) and the shelterwood designated not to be burned treatment (8%). No post-treatment significant differences were detected between treatments for scarlet oak basal area ($P=0.5959$) or scarlet oak as a percent of total basal area ($P=0.5908$). Scarlet oak basal area treatment means ranged from 0 to 1.34 square meters per hectare and percent of total basal area means ranged from 0 to 5 percent.

For sourwood basal area, no significant differences between treatments were found in either year 2000 ($P=0.7020$) or 2001 ($P=0.9592$). Treatment means ranged from 0.98 to 1.55 square meters per hectare of pre-treatment sourwood basal area while post-treatment basal area ranged from 0.88 to 1.19 square meters per hectare. No significant differences in the percentage of the total basal area comprised of sourwood were found in year 2000 ($P=0.8216$) or 2001 ($P=0.8529$). Pre-treatment means ranged from 4 to 8 percent and post-treatment means ranged from 4 to 8 percent of the total sourwood basal area.

For white oak basal area, year 2000, no significant differences were detected between treatments ($P=0.2165$). Treatment means ranged from 0.85 to 5.60 square meters of white oak per hectare. For white oak as a percent of total basal area for year 2000, treatment differences were also not significant ($P=0.1468$). Treatment means ranged from 3.26 to 18.18 percent. Post-treatment white oak basal area did not exhibit significant treatment differences ($P=0.7202$). Treatment means ranged from 2.05 to 4.98 square meters of white oak per hectare. For white oak as a percent of total basal area for year 2001 treatment differences ($P=0.8291$) were not significant. Treatment means ranged from 11.02 to 19.9 percent of the total basal area.

Pre-treatment yellow-poplar basal area exhibited no significant differences between treatments ($P=0.8393$). Treatment means ranged from 2.15 to 5.04 square meters of yellow-poplar per hectare. No significant differences between treatments ($P=0.7344$) were found in yellow-poplar as a percent of total basal area for year 2000. Treatment means ranged from 7.49 to 19.59 percent. Post-treatment yellow-poplar basal area did not significantly differ between treatments ($P=0.5671$). Treatment means ranged from 1.93 to

4.93 square meters of yellow-poplar per hectare. For yellow-poplar as a percent of total basal area for year 2001, treatment differences were not significant ($P=0.5380$). Yellow-poplar comprised 9 to 25 percent of the total basal area.

No significant treatment differences were detected for those minor species pooled for analysis in basal area ($P=0.5330$) or in the percentage of the total basal area ($P=0.6142$) comprised of those species in year 2000. Basal area treatment means ranged from 0.82 to 3.08 square meters per hectare comprised of this group amounting to 3 to 11 percent of the total basal area. Post-treatment basal area of this species group did not significantly differ between treatments ($P=0.0636$) nor did the species group as a percentage of the total basal area ($P=0.0660$). Treatment means ranged from 2.22 to 6.53 square meters per hectare of basal area amounting to 9.74 to 30.08 percent of the total basal area.

Total basal area of all species was calculated from diameter measurements, was analyzed by treatment. No pre-treatment significant treatment differences were detected for total basal area ($P=0.5926$). Treatment means ranged from 23.13 to 27.77 square meters of basal area per hectare. Similar results occurred in 2001. No post-treatment significant treatment differences were detected for total basal area ($P=0.1014$). Treatment means ranged from 17.14 to 24.31 square meters of basal area per hectare (Figure 4). These results are very similar to those found using a prism to measure basal area, as reported on page 50.

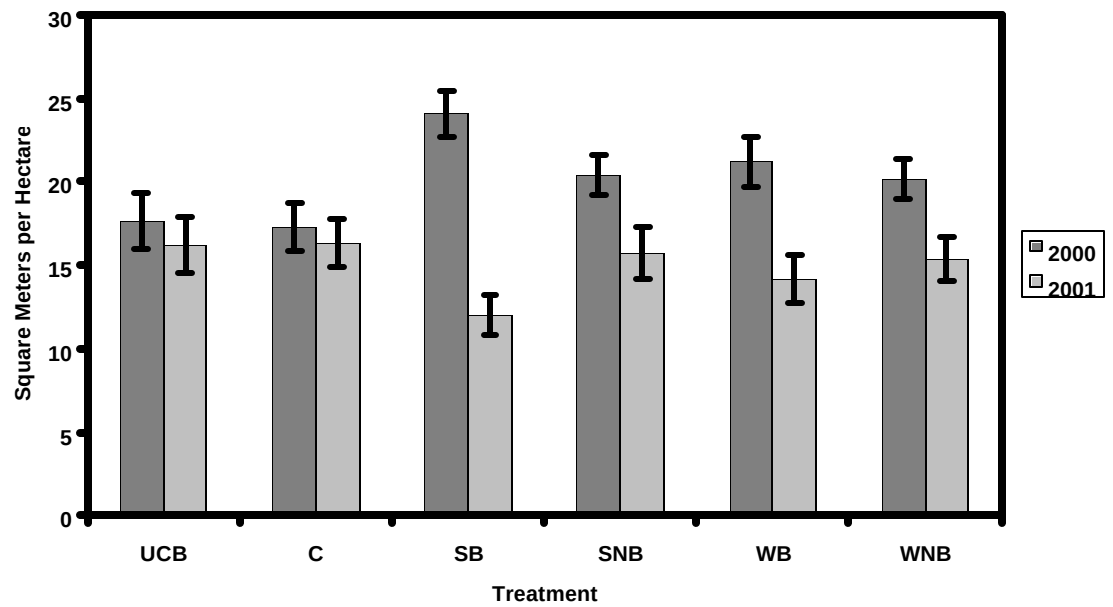


Figure 4. Total basal area calculated from overstory tree diameters by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

Sapling Analysis

This section of analysis represents the trees over 1.4-m tall but less than 4.6 cm in diameter breast height. Oaks and their competitors (stems per hectare) were analyzed individually. Also, the total number of stems of saplings per hectare was analyzed. Data in this section were analyzed within years to show significant differences. Very few saplings per hectare were found on the study sites. Much of the data was non-normal due to the sparse data values. The log-transformation of each variable was used, but often resulted in poor normality. Means shown are from the raw data. Species recorded are shown in Table 2.

Pre-treatment American beech saplings per hectare did not significantly differ between treatments ($P=0.2824$). Treatment means ranged from 29 to 404 American beech saplings per hectare. Post-treatment American beech saplings per hectare did not significantly differ between treatments ($P=0.2156$). Treatment means ranged from 0 to 141 American beech saplings per hectare.

Blackgum saplings per hectare did not significantly differ between treatments ($P=0.8473$) in pre-treatment year 2000. Treatment means ranged from 28 to 125 blackgum saplings per hectare. In year 2001, significant differences were detected between treatments (Figure 5). The wildlife thinnings, both burned and unburned (0 saplings/ha) had significantly fewer blackgum saplings than the shelterwood cut designated to be burned (37 saplings/ha).

No significant differences between treatments were found in chestnut oak saplings per hectare in year 2000 ($P=0.9689$) or year 2001 ($P=0.5937$) Pre-treatment means

Table 2. Sapling species list.

Scientific Name	Common Name	Scientific Name	Common Name
<i>Acer rubrum</i>	Red Maple	<i>Magnolia tripetala</i>	Umbrella Magnolia
<i>Acer saccharum</i>	Sugar Maple	<i>Morus rubra</i>	Red Mulberry
<i>Amelanchier arborea</i>	Serviceberry	<i>Nyssa sylvatica</i>	Blackgum
<i>Asimina triloba</i>	Pawpaw	<i>Oxydenrum arboreum</i>	Sourwood
<i>Carpinus caroliniana</i>	American Hornbeam	<i>Pinus strobus</i>	Eastern White Pine
<i>Carya glabra</i>	Pignut Hickory	<i>Prunus serotina</i>	Black Cherry
<i>Carya ovata</i>	Shagbark Hickory	<i>Quercus alba</i>	White Oak
<i>Carya tomentosa</i>	Mockernut Hickory	<i>Quercus montana</i>	Chestnut Oak
<i>Castanea dentata</i>	American Chestnut	<i>Quercus rubra</i>	Northern Red Oak
<i>Cercis canadensis</i>	Eastern Redbud	<i>Rhamnus caroliniana</i>	Carolina Buckthorn
<i>Cornus florida</i>	Flowering Dogwood	<i>Rhododenron maximum</i>	Great Rhododendron
<i>Fagus grandifolia</i>	American Beech	<i>Sassafras albidum</i>	Sassafras
<i>Fraxinus americana</i>	White Ash	<i>Tilia americana</i>	American Basswood
<i>Juniperus virginiana</i>	Eastern Redcedar	<i>Ulmus americana</i>	American Elm
<i>Lindera benzoin</i>	Spicebush	<i>Ulmus rubra</i>	Sliperry Elm
<i>Liriodendron tulipifera</i>	Yellow-Poplar	<i>Viburnum acerifolium</i>	Maple Leafed Viburnum
<i>Magnolia acuminata</i>	Cucumber Magnolia		

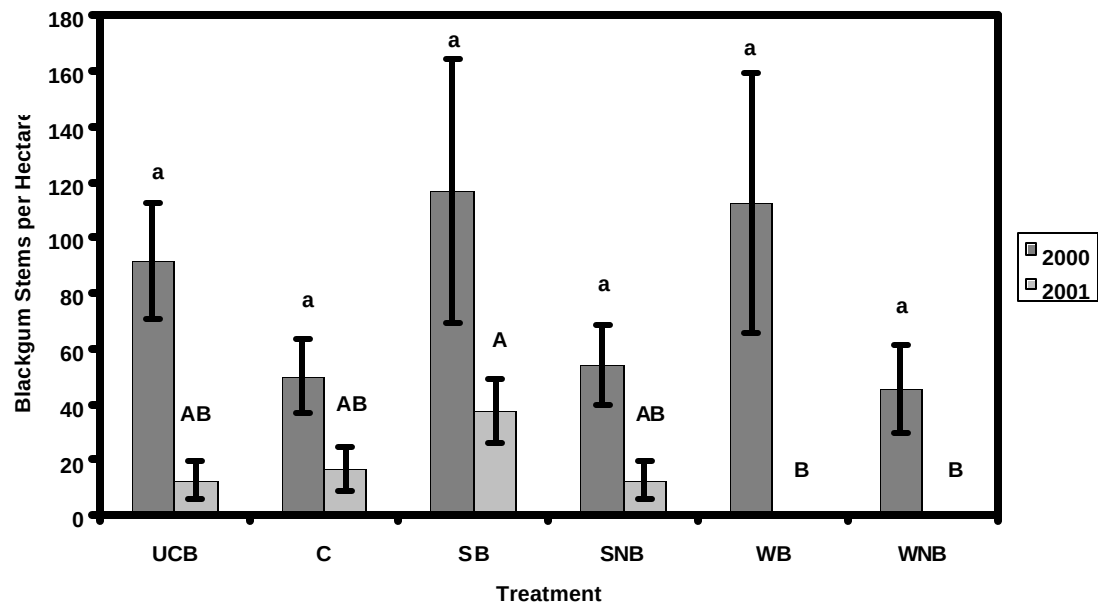


Figure 5. Blackgum saplings by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

ranged from 0 to 4 chestnut oak saplings per hectare while post-treatment means ranged from 0 to 17 chestnut oak saplings per hectare.

Pre-treatment flowering dogwood saplings per hectare did not differ significantly between treatments ($P=0.2882$). Treatment means ranged from 192 to 354 flowering dogwood saplings per hectare. Flowering dogwood saplings per hectare did not differ significantly between treatments ($P=0.1734$) in the post-treatment data either. Treatment means ranged from 12 to 158 flowering dogwood saplings per hectare.

No significant differences were found in hickory saplings per hectare between treatments in year 2000 ($P=0.7545$) or 2001 ($P=0.4596$). Pre-treatment means ranged from 12.5 to 96 hickory saplings per hectare. Post-treatment means ranged from 0 to 46 hickory saplings per hectare.

Pre-treatment red maple saplings per hectare did not differ significantly between treatments ($P=0.6454$). Treatment means ranged from 312 to 1,087 red maple saplings per hectare. No significant differences were detected between treatments ($P=0.0900$) in post-treatment year 2001 red maple saplings per hectare. Treatment means ranged from 108 to 354 saplings per hectare.

Red oak saplings per hectare did not significantly differ between treatments in either 2000 ($P=0.4823$) or 2001 ($P=0.1642$). Pre-treatment means ranged from 0 to 42 red oak saplings per hectare while post-treatment means ranged from 0 to 25 saplings per hectare.

No significant differences between treatments ($P=0.4756$) were detected for pre-treatment sassafras saplings per hectare. Treatment means ranged from 0 to 167 sassafras saplings per hectare. Post-treatment sassafras saplings per hectare did not significantly

differ between treatments ($P=0.2706$). Treatment means ranged from 0 to 92 saplings per hectare.

Sourwood saplings per hectare did not differ significantly between treatments in either 2000 ($P=0.5345$) or 2001 ($P=0.6106$). Pre-treatment means ranged from 96 to 221 sourwood saplings per hectare while post-treatment means ranged from 12 to 108 sourwood saplings per hectare.

No significant treatment differences in year 2000 ($P=0.5276$) or year 2001 ($P=0.1320$) sugar maple saplings per hectare were detected. Pre-treatment means ranged from 4 to 21 sugar maple saplings per hectare. Post-treatment means ranged from 0 to 8 saplings per hectare.

For pre-treatment white oak saplings per hectare, no significant treatment differences ($P=0.5151$) were detected. Treatment means ranged from 0 to 121 sugar maple saplings per hectare. Post-treatment white oak saplings per hectare showed similar results. No significant treatment differences ($P=0.4509$) were detected. Treatment means ranged from 0 to 75 white oak saplings per hectare.

No significant treatment differences were detected in year 2000 or 2001 yellow-poplar saplings per hectare. Pre-treatment means ranged from 4 to 171 yellow-poplar saplings per hectare while post-treatment means ranged from 0 to 112 yellow-poplar saplings per hectare.

For the pre-treatment total number of saplings, of all species, significant differences were found between treatments ($P=0.0251$). The shelterwood treatment designated to remain unburned (2,341 stems/ha) had significantly more saplings per hectare than the uncut burned treatment (1,346 stems/ha). Significant differences between

treatments were also detected in the post-treatment total number of saplings (Figure 6). The prescribed burned wildlife thinning (254 stems/ha) had significantly fewer saplings per hectare than the control (1,046 stems/ha) or shelterwood designated to be burned (725 stems/ha). Saplings were most significantly reduced in any treatment involving site disturbance, while prescribed burning also reduce the number of saplings per hectare, though not as dramatically.

Canopy Cover Analysis

For year 2000, pretreatment differences in canopy cover were not found between cells representing future treatments ($P=0.3809$). Pretreatment canopy cover means ranged from 96.87 to 97.35%. Significant differences in canopy cover were found between treatments ($P=0.0001$) in year 2001 (Figure 7). The control had significantly higher canopy cover (97.39%) than the prescribed burned wildlife thinning (89.16%) and the shelterwood cuts, both unburned and to be burned (85.88 and 86.70%). The uncut burned treatment had significantly higher canopy cover (94.28%) than both shelterwood cut treatments. The unburned wildlife thinning (92.77%) had significantly higher average cover than the shelterwood cut to be burned (85.88%).

Site Characteristics Analyses

No significant differences in percent slope were found between treatments ($P=0.8847$). Stand slopes ranged from 24 to 30 percent and treatment slope means ranged from 24 to 28 percent.

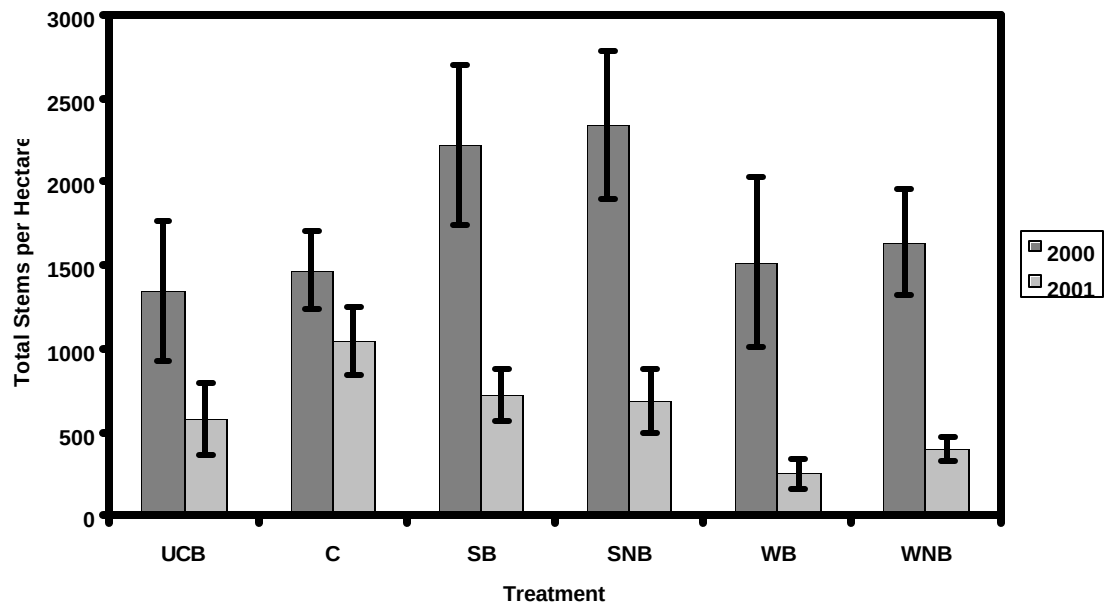


Figure 6. Total saplings by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

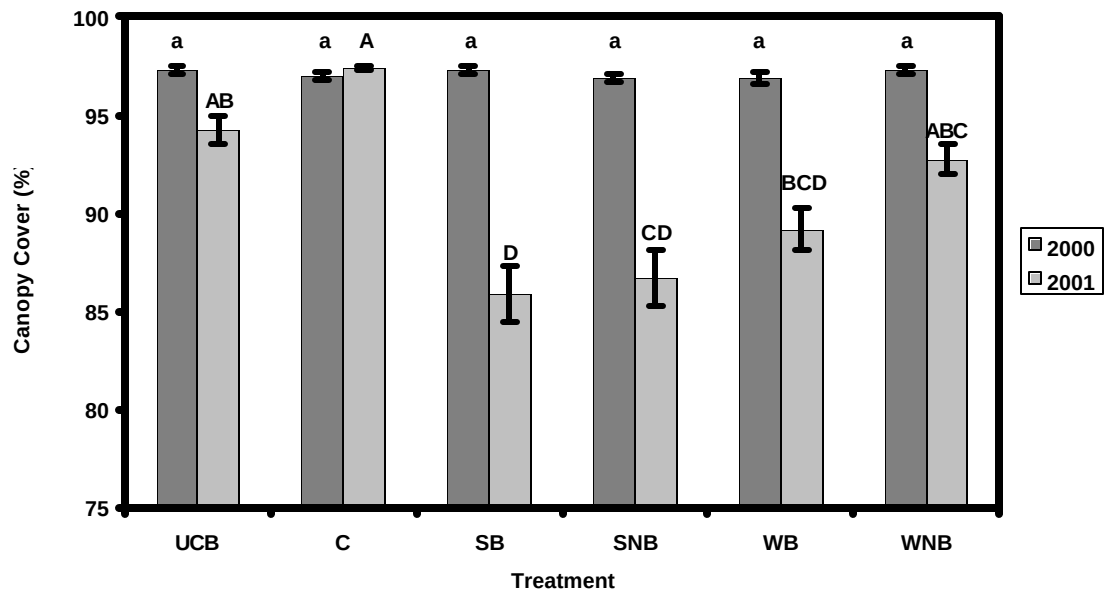


Figure 7. Percent canopy cover by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

In 2000, there were no significant differences in percent shrub crown cover between future treatment cells ($P=0.3984$) and means of these treatments ranged from 17.29 to 31.63 percent. In 2001, there were significant differences in percent shrub crown cover between treatments ($P=0.0035$, Figure 8). The uncut prescribed burned treatment had significantly less shrub cover (0%) than the control (21.04%) and shelterwood cuts (13.75 and 15.63%). The prescribed burned wildlife thinning had significantly less shrub cover (3.6%) than the control (0%). Prescribed burning significantly reduced shrub cover in the stands. Shrub cover data for year 2001 were slightly non-normal but was analyzed without transformation ($W=0.885$).

No significant differences in the pre-treatment year, 2000, basal areas measured with a hand-held prism were found between future treatments. Treatment means ranged from 20.20 to 23.41 square meters per hectare. In 2001, post-treatment, significant differences in basal area were found between treatments ($P=0.0001$, Figure 9). The control had significantly higher basal area ($25.02\text{-m}^2/\text{ha}$) than the unburned wildlife thinning ($20.43\text{-m}^2/\text{ha}$), the prescribed burned wildlife thinning ($19.51\text{-m}^2/\text{ha}$), and the shelterwood cuts, both unburned and to be burned (14 and $15.61\text{-m}^2/\text{ha}$). The uncut burned treatment ($23.64\text{-m}^2/\text{ha}$) and the unburned wildlife thinning had significantly higher mean basal areas than the shelterwood cuts. Basal area was significantly reduced in treatments where overstory removal was the primary goal. However, shelterwood cuts exhibited less basal area than wildlife thinnings.

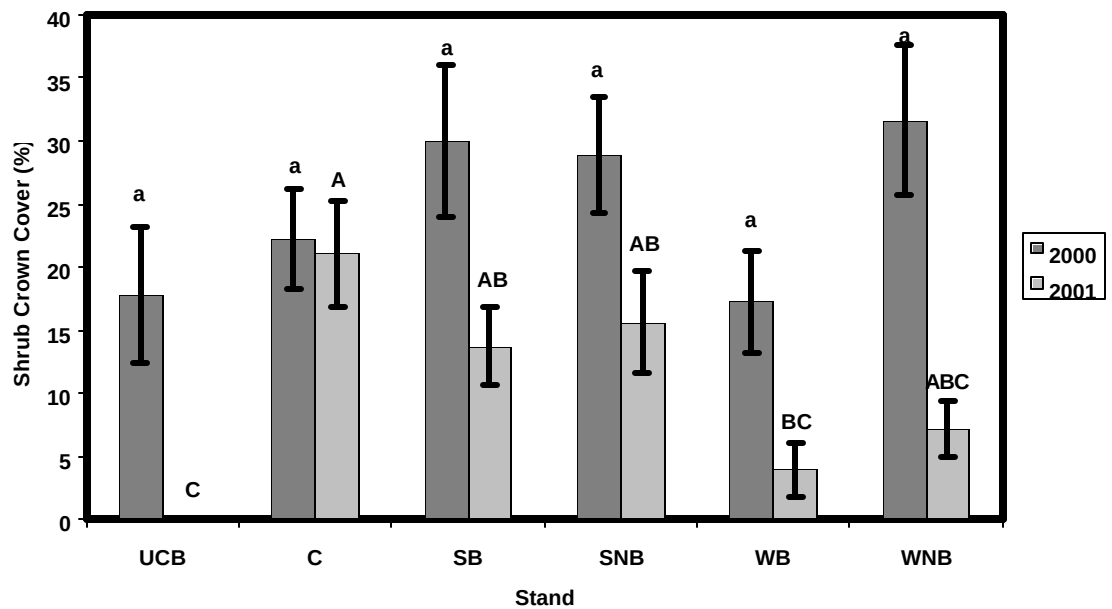


Figure 8. Percent shrub crown cover by treatment. Error bars and significance as found in Figure 3. Treatment codes as in Figure 1.

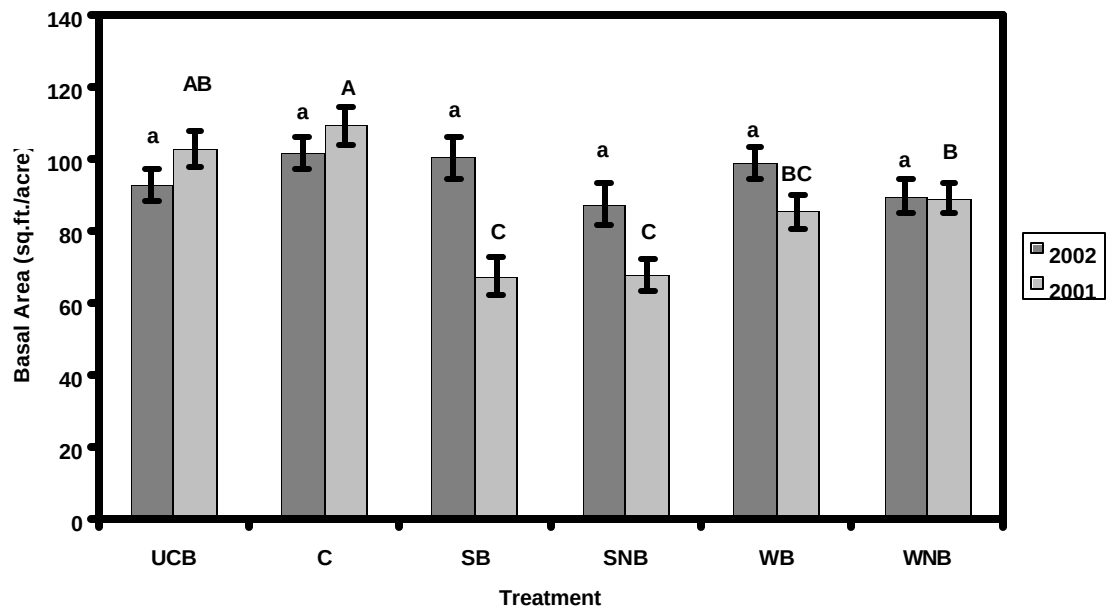


Figure 9 Prism basal area by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

Light and Soil Moisture Analyses

Due to non-normal distributions of the raw data, the analyses of PAR and soil moisture were conducted using the natural log of the original values. Means presented represent untransformed data. It is important to note that light and soil moisture data were collected only in the post-treatment year, 2001. The analysis of the PAR measurements taken at the 1-m height level showed significant differences in PAR between treatments ($P=0.0001$, Figure 10). The control had significantly lower PAR availability (3.8%) than all other treatments. Any cell that received a treatment exhibited increased levels of light (PAR) availability over the controls.

The light measurements taken at the 30.5-cm level showed similar results. Differences in PAR availability between treatments were very significant ($P=0.0001$, Figure 10). The control had significantly lower PAR (3.7%) at this level than any other treatment except the uncut burned treatment (8.2%). The uncut burned treatment had significantly lower light levels than the shelterwood treatment designated for burning (21%). In this analysis, any treatment involving overstory removal increased light (PAR) availability.

There were no significant differences in percent soil moisture between treatments ($P=0.2267$, Figure 11). Mean percent soil moisture ranged from 12.8 to 16.03 percent. However, in stand 2, when analyzed individually, the shelterwood treatments (11.6 and 14.1%) had significantly more soil moisture than the control (7.7%) or uncut burned treatment (8.9%, Figure 12).

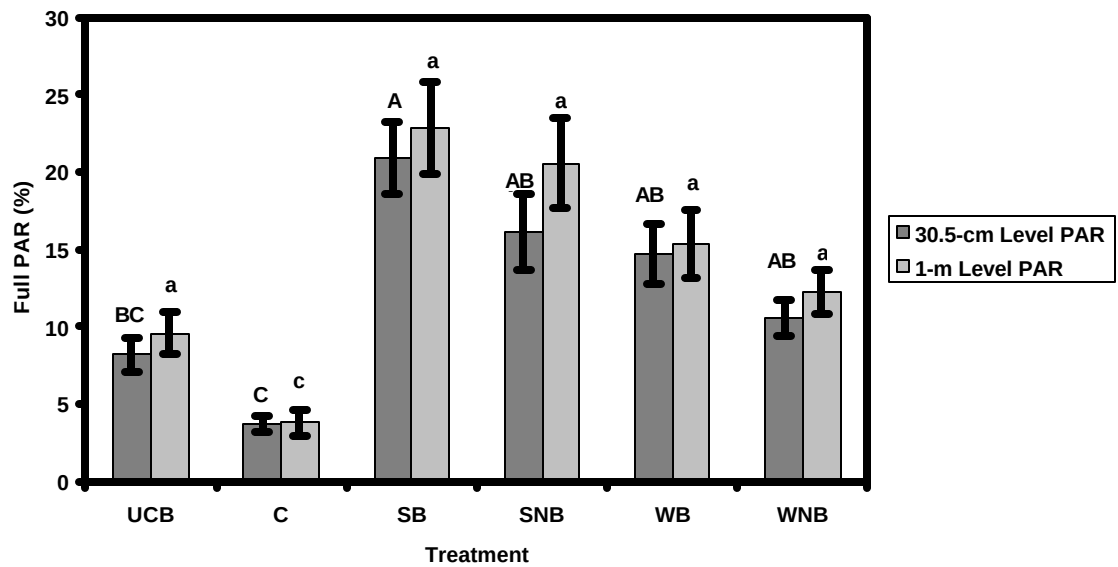


Figure 10. Percent of total PAR available by treatment for both the 30.5-cm and 1-m height measurements. Lower case letters were used for 1-m means and capital letters were used for the 30.5-cm means. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

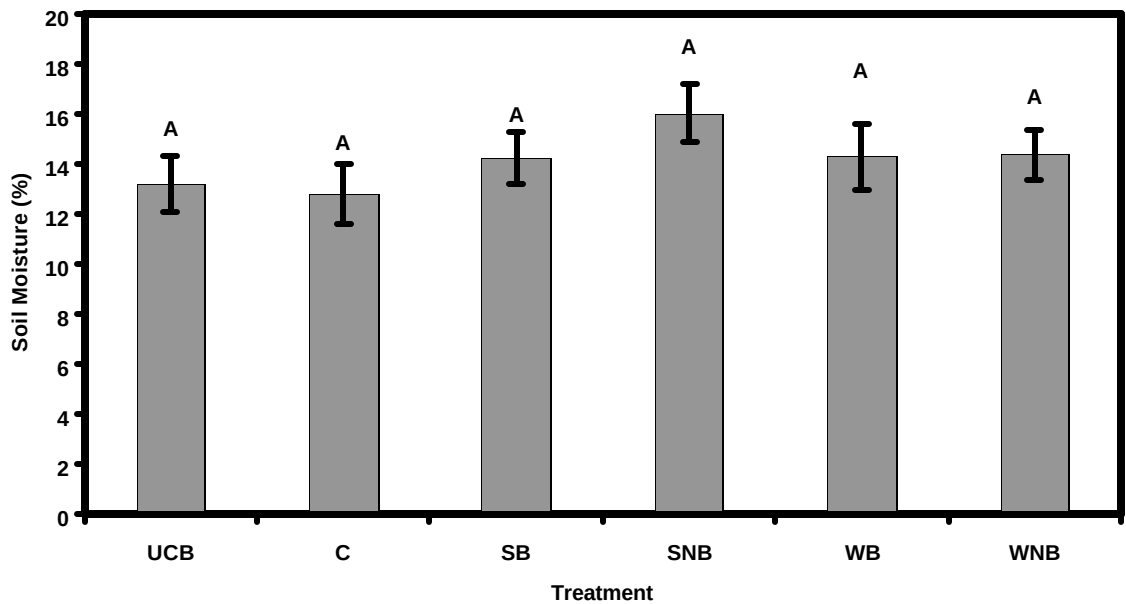


Figure 11. Percent soil moisture by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

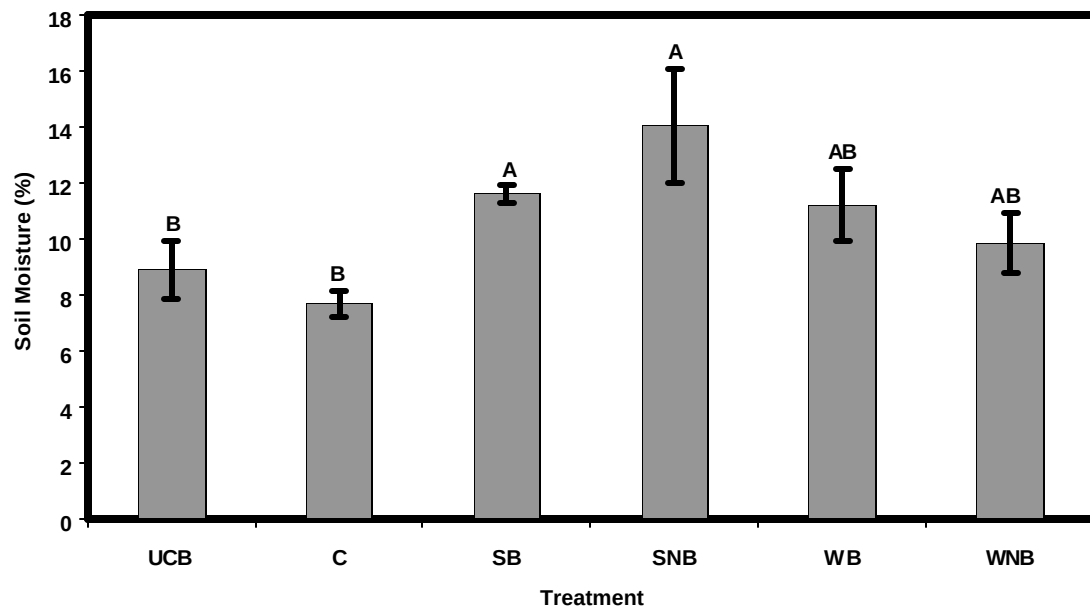


Figure 12. Percent soil moisture by treatment in stand 2. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

Herbaceous Cover Analysis

No significant differences in ground cover by herbaceous plants between cells assigned to future treatments were found ($P=0.2485$) prior to treatment. Treatment mean herbaceous cover ranged from 5.63 to 12.25%. In 2001, similar results were found. No significant differences in herbaceous cover were found between treatments ($P=0.0929$). Mean herbaceous cover ranged from 2.29 to 8.30% (Figure 13). The natural log of this variable was used in the statistical analysis due to non-normality in the raw data.

Before treatment in 2000, no significant differences in the average number of species per plot between treatments were found ($P=0.1902$). In 2001, post-treatment, no significant differences between treatments were found ($P=0.1774$, Figure 14). Although the number of species per plot was relatively low, many different species were found on the study area (Table 3).

In order to better illustrate the effects of the treatments on herbaceous cover, analysis was done in two, more thoroughly descriptive ways. The first was analysis based on major species groups of herbaceous plants and their specific ground coverage values. They were forbs, woody vines, graminoids (*Panicum spp.*), shrubs, and ferns.

Pre-treatment percent forb ground coverage was not significantly different between treatments ($P=0.0589$). Mean treatment differences ranged from 2.32 to 7.03 percent coverage. In post-treatment percent forb coverage, 2001, significant differences between treatments ($P=0.0116$) were found (Figure 15). The control had significantly higher percent forb coverage (4.59%) than the shelterwood cuts (0.88 and 0.94%). The natural log of this variable was used in this analysis due to non-normality in the data.

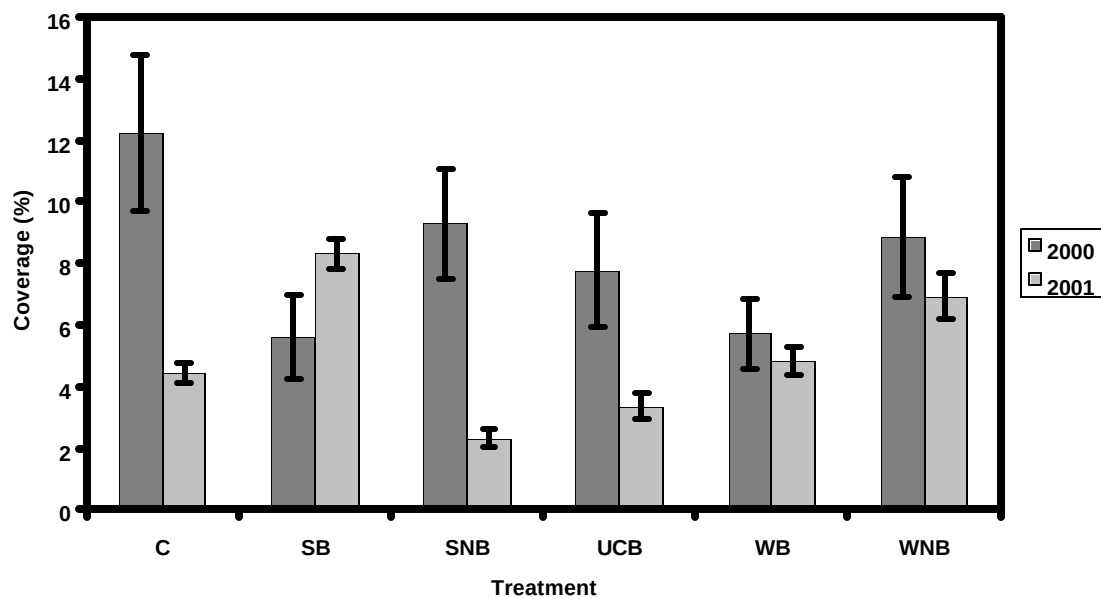


Figure 13. Percent cover of herbs by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

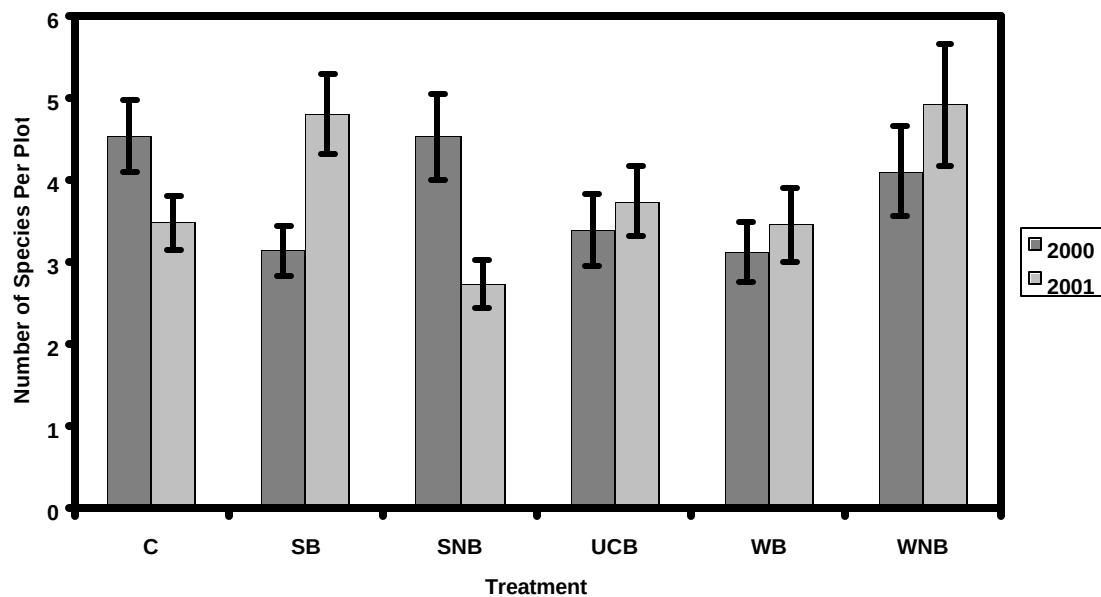


Figure 14. Herb richness per plot by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

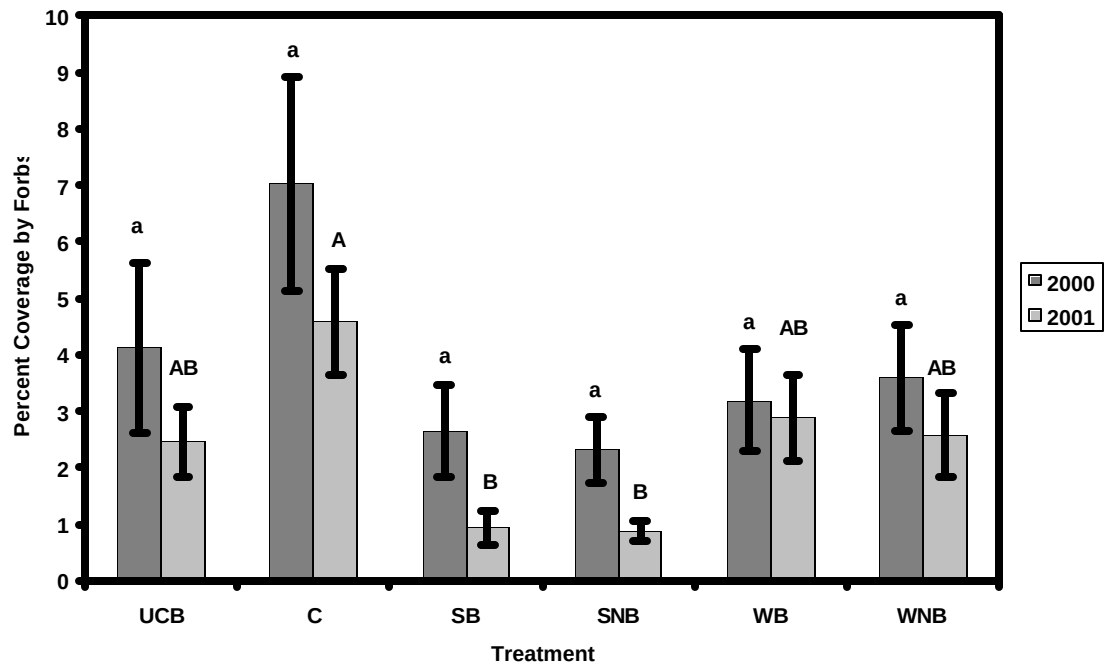


Figure 15. Percent cover comprised of forbs. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

Table 3. Herbaceous species list.

Scientific Name	Common Name	Scientific Name	Common Name
<i>Actaea pachypoda</i>	Doll's Eyes	<i>Hieracium</i> spp.	Hawkweed
<i>Adiantum pedatum</i>	Maidenhair Fern	<i>Hydrangea arborescens</i>	Wild Hydrangea
<i>Agrimonia pubescens</i>	Cocklebur	<i>Iris verna</i>	Dwarf Iris
<i>Agrimonia rostellata</i>	Cocklebur	<i>Liparis lilifolia</i>	Lily-leaved Twayblade
<i>Ambrosia</i> spp.	Ragweed	<i>Lonicera</i> spp.	Honeysuckle
<i>Amianthium muscaetoxicum</i>	Fly Poison	<i>Lycopodium obscurum</i>	Groundpine
<i>Amphicarpa bracteata</i>	Hogpeanut	<i>Lycopodium tristachyum</i>	Ground-cedar
<i>Anemone quinquefolia</i>	Wood Anemone	<i>Lycopus virginicus</i>	Bugleweed
<i>Angelica</i> spp.	Angelica	<i>Lysimachia quadrifolia</i>	Whorled Loosestrife
<i>Arisaema triphyllum</i>	Jack-in-the-Pulpit	<i>Medeola virginiana</i>	Indian Cucumber
<i>Aristolchia serpentaria</i>	Birthwort (Dutchman's Pipe)	<i>Menispermum canadense</i>	Moonseed
<i>Asplenium</i> spp.	Spleenwort	<i>Microstegium vimineum</i>	Japanese Grass
<i>Aster</i> spp.	Aster	<i>Mitchella repens</i>	Partridge Berry
<i>Athyrium asplenoides</i>	Southern Lady Fern	<i>Osmunda cinnamomea</i>	Cinnamon Fern
<i>Aureolaria</i> spp.	Aureolaria	<i>Oxalis</i> spp.	Wood Sorrel
<i>Botrychium virginianum</i>	Rattlesnake Fern	<i>Panax quinquefolium</i>	Ginseng
<i>Brachyelytrum erectum</i>	Beauvois	<i>Panicum</i> spp.	Panicum
<i>Chamaelirium luteum</i>	Blazing-star	<i>Parthenocissus quinquefolia</i>	Virginia Creeper
<i>Chimaphila maculata</i>	Pipsissewa Wintergreen	<i>Phryma leptostachya</i>	Lopseed
<i>Cimicifuga racemosa</i>	Black Cohosh	<i>Phytolacca americana</i>	Pokeweed
<i>Circaea lutetiana (canadensis)</i>	Enchanter's Nightshade	<i>Polygonatum biflorum</i>	Solomon's Seal
<i>Collinsonia</i> spp.	Collinsonia	<i>Polystichum acrostichoides</i>	Christmas Fern
<i>Conopholis americana</i>	Squawroot	<i>Prenanthes altissima</i>	Tall White Lettuce
<i>Cynoglossum virginianum</i>	Wild Comfrey	<i>Pteridium aquilinum</i>	Bracken Fern
<i>Cypripedium acaule</i>	Pink Lady Slipper	<i>Rosa</i> spp.	Rose
<i>Desmodium glutinosum</i>	Beggar's Lice	<i>Rosa</i> spp.	Rose
<i>Desmodium rotundifolium</i>	Dollarleaf	<i>Rubus</i> spp.	Raspberry
<i>Desmodium</i> spp.	Trefoil	<i>Sanicula gregaria</i>	Snakeroot
<i>Dioscorea villosa</i>	Wild Yam	<i>Sanicula</i> spp.	Snakeroot
<i>Diosporum</i> spp.	Mandarin	<i>Scutellaria elliptica</i>	Skullcap
<i>Dryopteris</i> spp.	Shield Fern	<i>Smilacina racemosa</i>	False Solomon's Seal
<i>Elephantopus</i> spp.	Elephant's Foot	<i>Smilax glauca</i>	Greenbrier
<i>Epifagus virginiana</i>	Beech Drops	<i>Solidago caesia</i>	Blue-stem Goldenrod
<i>Erechtites hieracifolia</i>	Fireweed	<i>Stellaria pubera</i>	Chickweed (Giant)
<i>Euonymus americanus</i>	Strawberry Bush	<i>Streptopus roseus</i>	Rosy Twisted-Stalk
<i>Eupatorium</i> spp.	Thoroughwort	<i>Thalictrum thalictroides</i>	Wind flower
<i>Galactia</i> spp.	Galactia	<i>Thaspium</i> spp.	Meadow Parsnip
<i>Galium circaeans</i>	Bedstraw	<i>Thelypteris hexagonoptera</i>	Broad Beech-Fern
<i>Galium triflorum</i>	Bedstraw	<i>Toxicodendron radicans</i>	Poison Ivy
<i>Gentiana</i> spp.	Gentian	<i>Uvularia perfoliata</i>	Bellwort
<i>Geranium maculatum</i>	Wild Geranium	<i>Uvularia sessilifolia</i>	Bellwort
<i>Geum virginianum</i>	Avens	<i>Viola palmata</i>	Early-Blue Violet
<i>Goodyera pubescens</i>	Downy Rattlesnake Plantain	<i>Viola</i> spp.	Violet
<i>Panicum</i> spp., <i>Carex</i> spp., <i>Etc.</i>	Graminoids	<i>Waldsteinia fragarioides</i>	Barren Strawberry
<i>Hexastylis arifolia</i>	Little Brown Jug	<i>Zizia aptera</i>	Meadow Parsnip

Pre-treatment percent woody vine coverage did not significantly differ between treatments ($P=0.0844$). Treatment means ranged from 1.32 to 3.24 percent. Percent ground coverage by woody vines, no post-treatment differences were found between treatments ($P=0.0693$). Treatment means ranged from 0.70 to 2.89 percent woody vine coverage. The natural log of this variable was used in this analysis due to non-normality in the data.

Grasses and sedges (Graminoids) were classified into one group due to the difficulty of identifying non-flowering or fruiting stems on the study area. Our measurements took place at the end of the growing season, when no flowering or fruiting structures were apparent, thus making it very difficult to separate the few graminoid species that were found. No pre-treatment significant differences were found between treatments ($P=0.1566$) in percent coverage by graminoids. Treatment means ranged from 0.12 to 1.17 percent coverage by graminoids. Post-treatment data, year 2001, also indicated no significant differences between treatments in percent coverage by graminoids ($P=0.8037$). Treatment means ranged from 0.06 to 0.29 percent graminoid coverage. The square root of this variable was used in the statistical analysis due to the non-normality of the data. The normality for year 2000 transformed graminoid coverage data was poor ($W=0.862$), however, since year 2001 transformed data were normally distributed ($W=0.906$), untransformed data were used for analysis.

No pre-treatment significant differences were found between treatments ($P=0.5065$) in percent ground coverage by shrubs. Treatment means ranged from 0.001 to 0.04 percent coverage by shrubs. Post-treatment data indicated similar results with no significant treatment differences ($P=0.1158$) detected. Treatment means varied from

0.001 to 0.03 percent coverage by shrubs. The arc sine transformation of this variable was used due to non-normality in the raw data. Even in light of the transformations, poor normality was a problem for both pre-treatment ($W=0.477$) and post-treatment ($W=0.629$) data.

No pre-treatment significant differences were found between treatments (.7342) in percent ground coverage by ferns. Treatment means ranged from 0.62 to 3.06 percent coverage by ferns. Post-treatment data, year 2001, showed no significant differences between treatments ($P=0.8102$) in percent coverage by ferns. Treatment means ranged from 0.60 to 1.30 percent coverage by ferns. The natural log of this variable was used due to non-normality in the raw data.

Another method of analysis that was used to assess the herbaceous data was analysis of the dominant species and their associated ground coverages. These species included beggar's lice (*Desmodium glutinosum*), Christmas fern (*Polystichum acrostichoides*), little brown jug (*Hexastylis arifolia*), poison ivy (*Toxicodendron radicans*), spotted wintergreen (*Chimaphila maculata*), and greenbrier (*Smilax spp.*).

No significant differences between treatments ($P=0.3352$) were found in pre-treatment percent ground coverage by beggar's lice. Treatment means ranged from 1.25 to 4.95 percent coverage by beggar's lice. Post-treatment percent coverage by beggar's lice data was similar. No significant differences between treatments ($P=0.1017$) were found. Treatment means ranged from 0.43 to 2.61 percent coverage by beggar's lice. The natural log of this variable was used for statistical analysis due to non-normality in the raw data.

Pre-treatment percent ground coverage by Christmas fern did not significantly

differ between treatments ($P=0.8297$). Treatment means ranged from 0.62 to 2.67 percent coverage by Christmas fern. The post-treatment data also indicated no significant differences in percent coverage by Christmas fern between treatments ($P=0.9450$). Treatment means ranged from 0.57 to 0.94 percent. The natural log of this variable was used for statistical analysis due to non-normality in the raw data.

Pre-treatment, 2000, data showed no significant differences between treatments ($P=0.8815$) in percent ground coverage by little brown jug. Treatment means ranged from 0.18 to 0.49 percent coverage by little brown jug. No significant differences between treatments ($P=0.0696$) were detected in post-treatment percent coverage by little brown jug. Treatment means ranged from 0.09 to 0.55 percent coverage. The natural log of this variable was used for statistical analysis due to non-normality in the data.

For pre-treatment percent ground coverage by poison ivy, no significant differences were found between treatments ($P=0.2712$). Treatment means ranged from 0.03 to 0.56 percent coverage by poison ivy. Post-treatment percent cover by poison ivy, year 2001, also showed no significant differences between treatments ($P=0.3176$). Treatment means ranged from 0.02 to 0.62 percent coverage. The arc sine of this variable was used for statistical analysis due to non-normality of the raw data. Even so, both pre and post-treatment transformed data had poor normality ($W=0.7846$ and $W=0.5499$).

No significant differences were detected between treatments in percent ground coverage by spotted wintergreen in either year 2000, pre-treatment ($P=0.8177$), or 2001, post-treatment ($P=0.1484$). Year 2000 treatment means ranged from 0.08 to 0.16 percent coverage by wintergreen while year 2001 treatment means ranged from 0.01 to 0.05 percent coverage by wintergreen. The square root of this variable was used for statistical

analysis due to the non-normality of the raw data.

Percent ground coverage by greenbrier did not significantly differ between treatments ($P=0.5469$) in 2000, before treatments. Treatment means ranged from 1.17 to 1.52 percent coverage by greenbrier. No significant differences were found between treatments ($P=0.0717$) in post-treatment percent coverage by greenbrier data. Treatment means ranged from 0.59 to 1.19 percent coverage by greenbrier.

Woody Regeneration Analysis for Stems <10-cm

The regeneration of several key woody species <10-cm tall (stems per hectare), including oaks and their competitors, were analyzed individually. Also, the total number of stems of woody regeneration per hectare was analyzed. Data in this section were analyzed by year to show significant differences. All of the raw data analyzed here appeared to be non-normal. Analysis was conducted using the log-transformation of each variable with raw means presented in graphics. Table 4 lists the species recorded in this size class.

Pre-treatment American beech stems per hectare <10-cm tall did not significantly differ between treatments ($P=0.6056$). Treatment means ranged from 41 to 153 stems per hectare. Post-treatment American beech stems per hectare <10-cm tall also did not significantly differ between treatments ($P=0.2568$). Treatment means ranged from 0 to 409 stems per hectare of American beech. Non-normality of American beech stems per hectare <10-cm tall could not be corrected through transformation.

No significant differences were found in black cherry stems per hectare <10-cm tall for pre-treatment or post-treatment years. Pre-treatment means ranged from 205

Table 4. Woody regeneration species list.

Scientific Name	Common Name	Scientific Name	Common Name
<i>Acer rubrum</i>	Red Maple	<i>Oxydenrum arboreum</i>	Sourwood
<i>Acer saccharum</i>	Sugar Maple	<i>Pinus echinata</i>	Shortleaf Pine
<i>Amelanchier arborea</i>	Serviceberry	<i>Pinus strobus</i>	Eastern White Pine
<i>Aralia spinosa</i>	Devil's Walking Stick	<i>Prunus serotina</i>	Black Cherry
<i>Asimina triloba</i>	Pawpaw	<i>Quercus alba</i>	White Oak
<i>Carpinus caroliniana</i>	American Hornbeam	<i>Quercus montana</i>	Chestnut Oak
<i>Carya spp.</i>	Hickory	<i>Quercus spp.</i>	Red Oaks
<i>Castanea dentata</i>	American Chestnut	<i>Rhamnus caroliniana</i>	Carolina Buckthorn
<i>Cercis canadensis</i>	Eastern Redbud	<i>Rhododendron spp.</i>	Azalea
<i>Cornus florida</i>	Flowering Dogwood	<i>Rhus copallina</i>	Smooth Sumac
<i>Diospyros virginiana</i>	Common Persimmon	<i>Rosa spp.</i>	Rose Vine
<i>Fagus grandifolia</i>	American Beech	<i>Sambucus canadensis</i>	Elderberry
<i>Fraxinus americana</i>	White Ash	<i>Sassafras albidum</i>	Sassafras
<i>Illex opaca</i>	American Holly	<i>Tilia americana</i>	American Basswood
<i>Juniperus virginiana</i>	Eastern Redcedar	<i>Tsuga canadensis</i>	Eastern Hemlock
<i>Liriodendron tulipifera</i>	Yellow-Poplar	<i>Ulmus americana</i>	American Elm
<i>Magnolia acuminata</i>	Cucumber Magnolia	<i>Vaccinium spp.</i>	Blueberry
<i>Magnolia tripetala</i>	Umbrella Magnolia	<i>Viburnum acerifolium</i>	Maple Leafed Viburnum
<i>Nyssa sylvatica</i>	Blackgum	<i>Vitis spp.</i>	Grapevine

to 573 black cherry stems per hectare while post-treatment means ranged from 214 to 297 black cherry stems per hectare.

No pre-treatment significant treatment differences were found for blackgum stems per hectare <10-cm tall ($P=0.6580$). Treatment means ranged from 379 to 573 blackgum stems per hectare. Post-treatment data showed significant differences between treatments ($P=0.0008$) in blackgum stems per hectare <10-cm tall (Figure 16). The prescribed burned wildlife thinning (2160 stems/ha) had significantly more blackgum stems <10-cm tall than both shelterwood cuts (696 and 1044 stems/ha), the unburned wildlife thinning (880 stems/ha), and the control (798 stems/ha). The uncut burned treatment (1,638 stems/ha) had significantly more blackgum stems <10-cm tall than the unburned wildlife thinning and the control.

No significant differences in blueberry stems per hectare <10-cm tall were detected in either pre-treatment or post-treatment data. Pre-treatment means ranged from 1945 to 3101 blueberry stems per hectare while post-treatment means ranged from 2,303 to 13,124 blueberry stems per hectare.

Pre-treatment chestnut oak stems per hectare <10-cm tall did not significantly differ between treatments ($P=0.0619$). Treatment means ranged from 819 to 1,771 chestnut oak stems per hectare. Post-treatment differences between treatments were significant ($P=0.0334$) in chestnut oak stems per hectare <10-cm tall (Figure 17). The Tukey's mean separation showed that the prescribed burned wildlife thinning (2,600 stems/ha) had significantly more chestnut oak stems than the shelterwood cut not to be burned (921 stems/ha). The raw means indicated that the control (3,286 stems/ha)

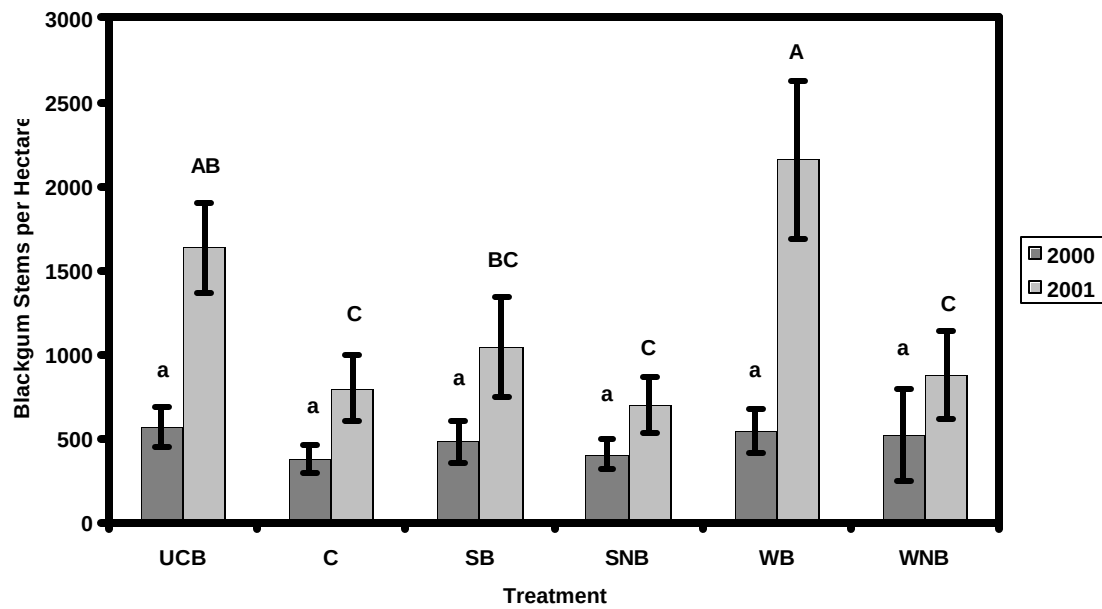


Figure 16. Blackgum regeneration <10-cm tall by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

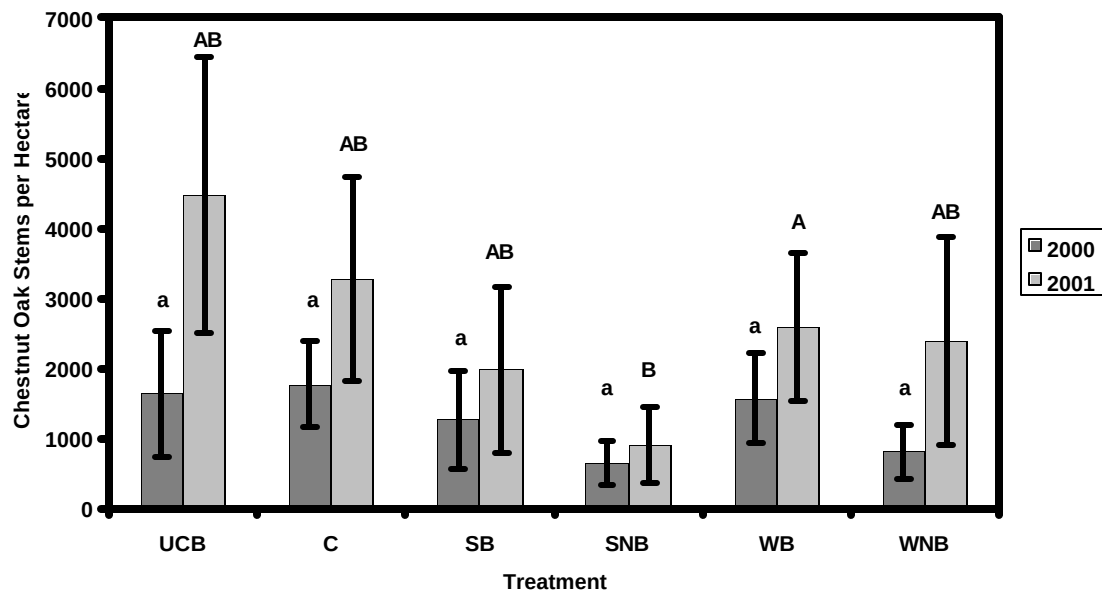


Figure 17. Chestnut oak regeneration <10-cm tall by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

and the uncut burned areas (4,484 stems/ha) had more chestnut oak stems than the prescribed burned wildlife thinning. The data were highly variable, with large standard errors.

Pre-treatment differences between treatments in flowering dogwood stems per hectare <10-cm tall were marginally significant (.0784). The prescribed burned wildlife thinning (143 stems/ha) had significantly fewer flowering dogwood stems than the shelterwood cut designated not to be burned (499 stems/ha). No post-treatment significant differences were found in flowering dogwood stems per hectare <10-cm tall ($P=0.0986$). Treatment means ranged from 112 to 471 flowering dogwood stems per hectare.

The overall pre-treatment analysis for hickory stems per hectare <10-cm tall, was not significant ($P=0.4338$). However, significant differences between treatments ($P=0.0170$) were found in the analysis and Tukey's mean separation indicated significant differences in hickory stems per hectare between treatments. The uncut burned treatment (72 stems/ha) had significantly fewer hickory stems per hectare than the shelterwood cut designated to be burned (184 stems/ha). Post-treatment differences between treatments were significant ($P=0.0230$, Figure 18). Although significant in the model, the Tukey's mean separation indicated no treatment differences. Treatment means ranged from 133 to 532 hickory stems per hectare <10-cm tall. Prescribed burned areas had substantially more hickory than unburned areas.

Pre-treatment red maple stems per hectare <10-cm tall did not significantly differ between treatments ($P=0.6042$). Treatment means ranged from 18,970 to 29,822 red maple stems per hectare. Post-treatment red maple stems per hectare <10-cm tall

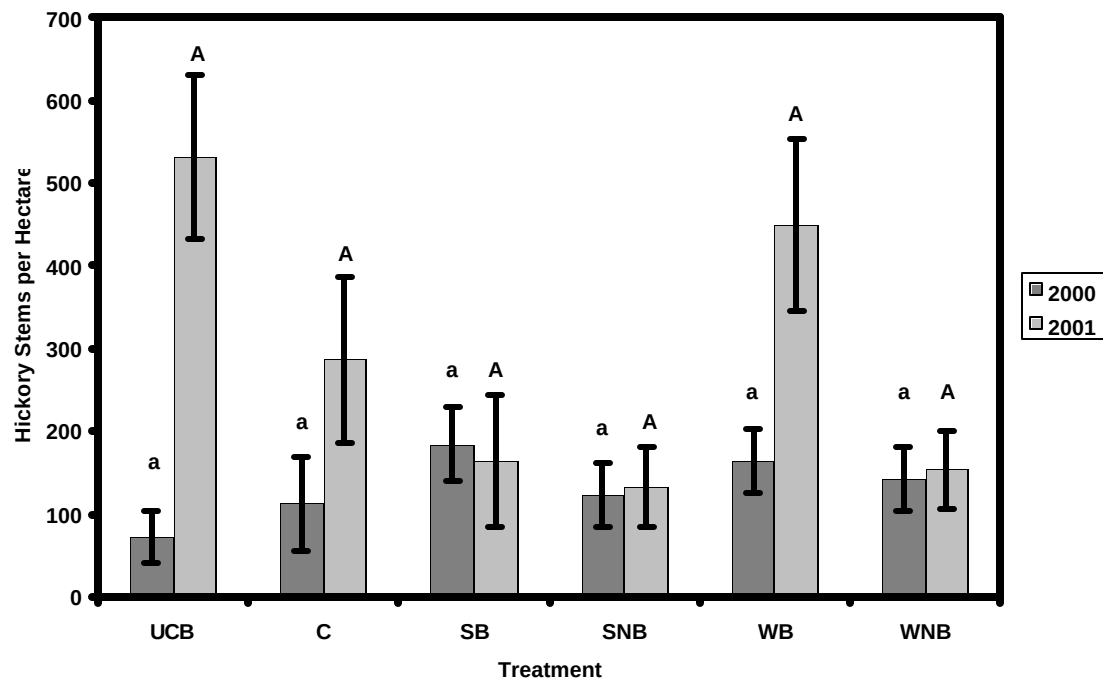


Figure 18. Hickory regeneration <10-cm tall by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

also did not significantly differ between treatments ($P=0.0879$, Figure 19). Treatment means ranged from 11,538 to 26,648 red maple stems per hectare <10-cm tall. When comparing the two years, any treatment involving site disturbance or prescribed burning produced reductions in red maple density, while the control saw an increase in red maple.

Red oaks, including northern red, black, and scarlet oaks, were analyzed as a group. For pre-treatment red oak stems per hectare <10-cm tall, no significant differences between treatments ($P=0.1776$) were found. Treatment means ranged from 113 to 379 red oak stems per hectare. Red oak stems per hectare <10-cm tall, post-treatment, did not significantly differ between treatments ($P=0.1795$). Treatment means ranged from 122 to 1,064 red oak stems per hectare <10-cm tall. However, the means, though not significant, indicate differences between the burned and unburned treatments. In overall red oaks, a slight increase is shown in the uncut burned treatment when compared to pretreatment levels (Figure 20). In stand 3, the effect of burning and overstory removal is more pronounced. This stand showed significant differences between treatments in red oaks (Figure 21). This also occurred dramatically in stand 1 red oaks.

No significant differences between treatments ($P=0.0777$) were found in pre-treatment sassafras stems per hectare <10-cm tall. Treatment means ranged from 389 to 1,136 sassafras stems per hectare. Significant post-treatment differences between treatments ($P=0.0001$) in sassafras stems per hectare <10-cm tall were found (Figure 22). The uncut burned treatment (23,085 stems/ha) and prescribed burned wildlife thinning (47,911 stems/ha) had significantly more sassafras stems per hectare than any other treatment.

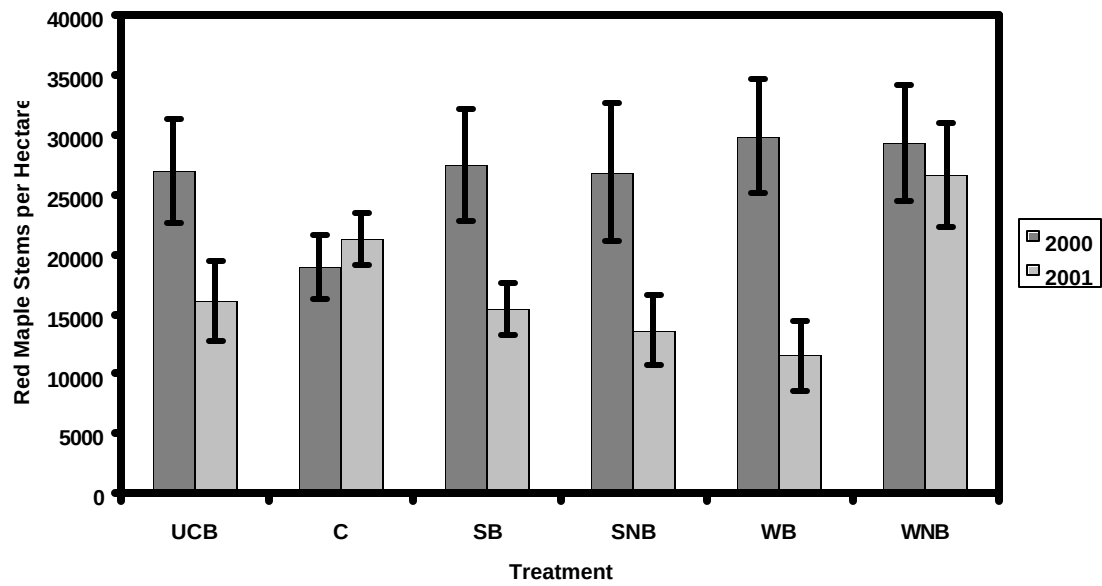


Figure 19. Red maple regeneration <10-cm tall by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

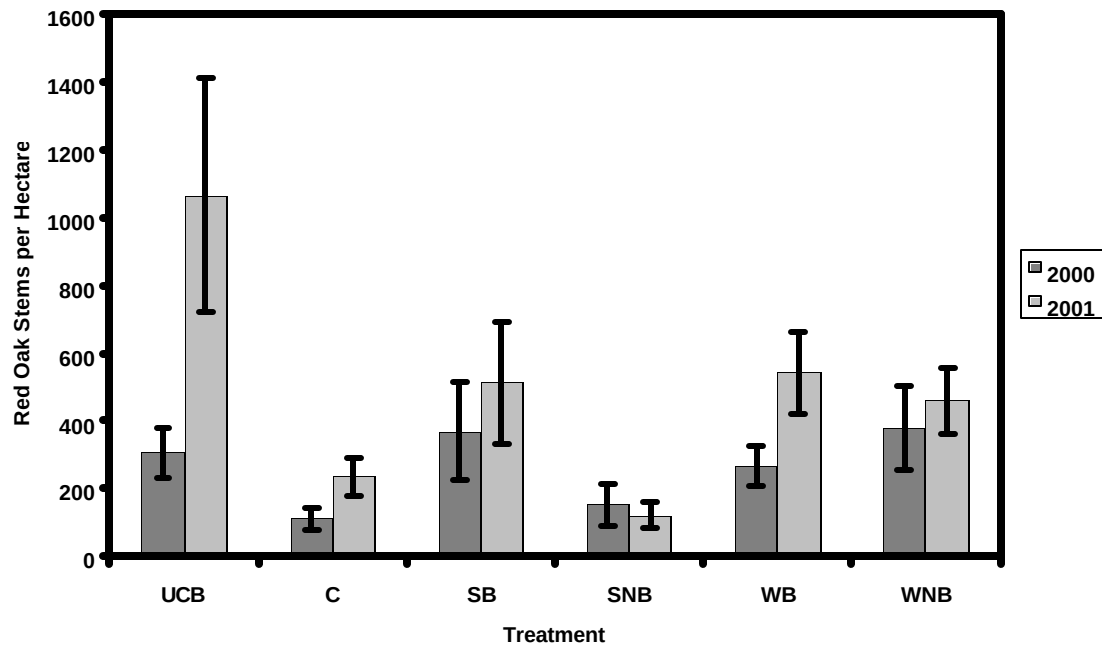


Figure 20. Red oak regeneration <10-cm tall by treatment Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

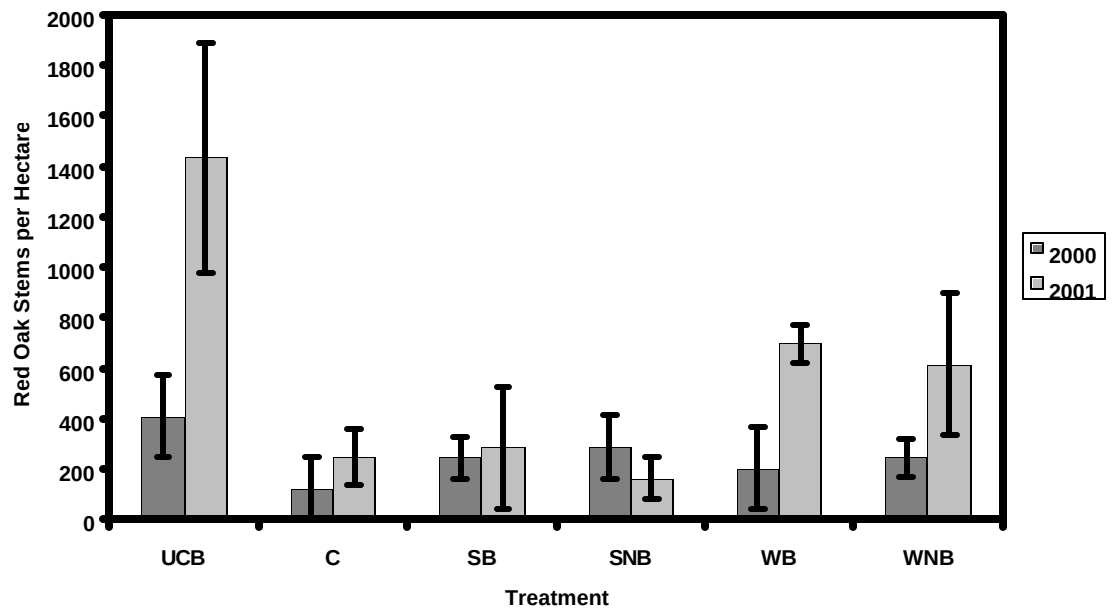


Figure 21. Red oak regeneration <10-cm tall by treatment in stand 3. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

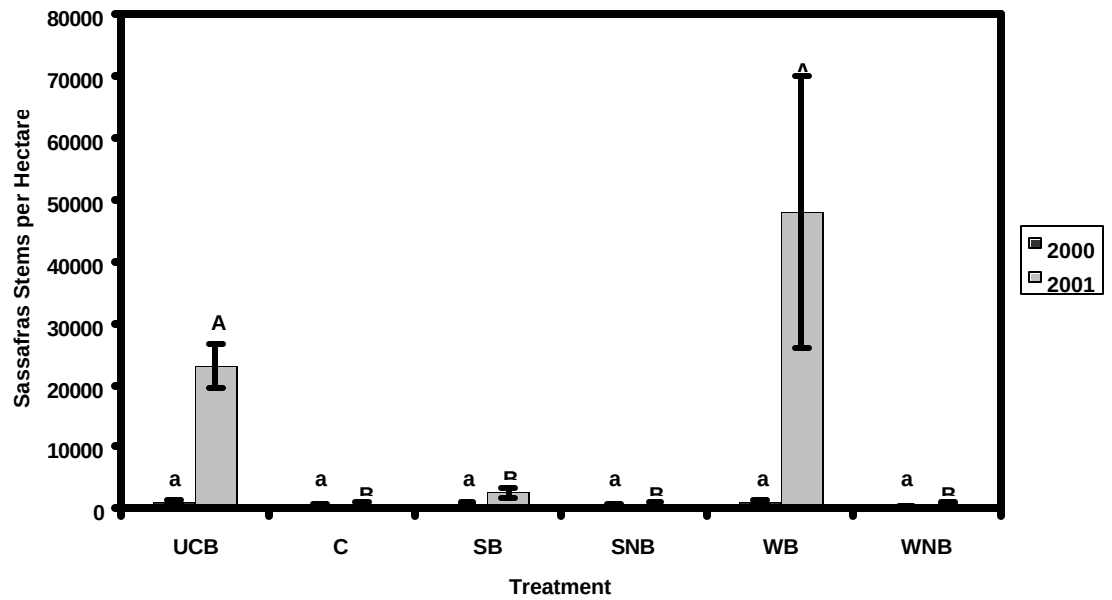


Figure 22. Sassafras regeneration <10-cm tall by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

Pre-treatment sugar maple stems per hectare <10-cm tall did not significantly differ between treatments ($P=0.4876$). Treatment means ranged from 72 to 594 sugar maple stems per hectare. No significant treatment differences ($P=0.9821$) were found in post-treatment sugar maple stems per hectare <10-cm tall data. Treatment means ranged from 194 to 952 sugar maple stems per hectare.

No significant pre-treatment differences were detected between treatments ($P=0.7503$) for white oak stems per hectare <10-cm tall. Treatment means ranged from 194 to 1,259 white oak stems per hectare. No significant treatment differences were found in post-treatment white oak stems per hectare <10-cm tall. Treatment means ranged from 184 to 6,848 white oak stems per hectare (Figure 23). White oak stems in prescribed burned treatments had almost 9 times the amount of stems <10-cm tall than any other treatment. Stand 1 showed significant differences between treatments in white oaks when analyzed individually (Figure 24). This also occurred in stand 2.

Treatments did not significantly differ ($P=0.4400$) in pre-treatment yellow-poplar stems per hectare <10-cm tall. Treatment means ranged from 123 to 553 yellow-poplar stems per hectare. Post-treatment differences between treatments were significant ($P=0.0041$) in yellow-poplar stems per hectare <10-cm tall (Figure 25). The prescribed burned wildlife thinning (25,859 stem/ha) and the uncut burned treatment (31,664 stems/ha) had significantly more yellow-poplar stems per hectare than the shelterwood cut to remain unburned (307 stems/ha) and the control (184 stems/ha). The raw means indicate that the shelterwood cut designated to be burned (225 stems/ha) may have significantly fewer yellow-poplar stems per hectare than the prescribed burned areas.

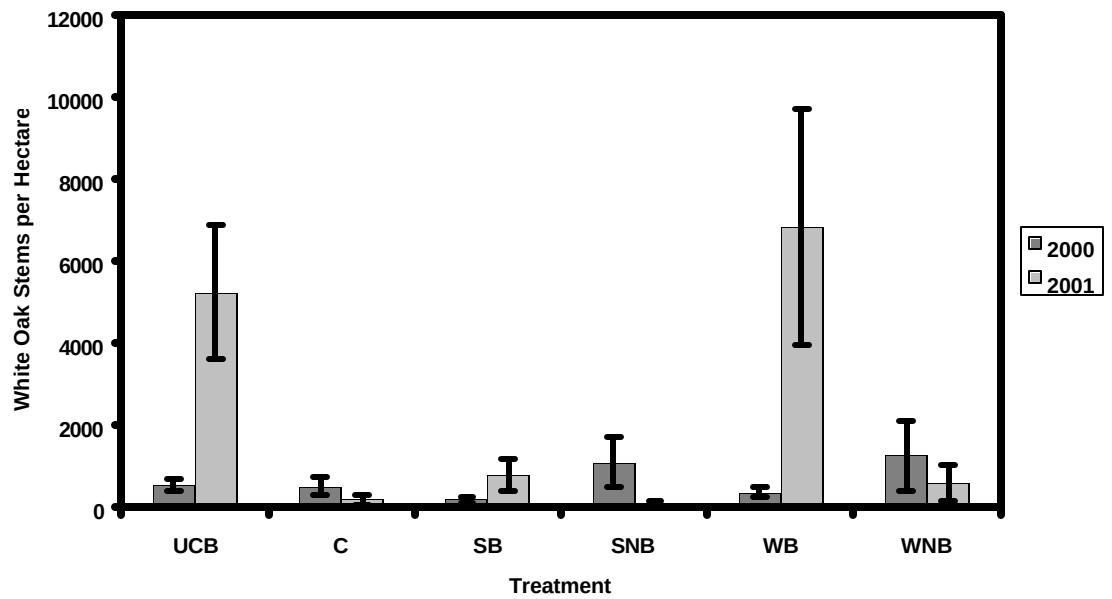


Figure 23. White oak regeneration <10-cm tall by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

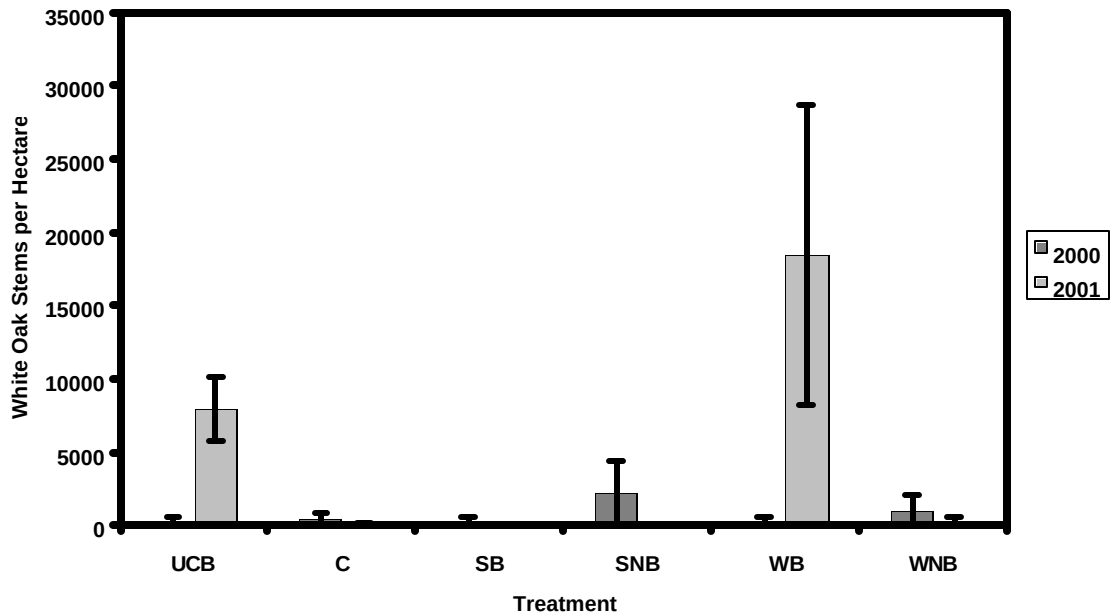


Figure 24. White oak regeneration <10-cm tall by treatment in stand 1. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

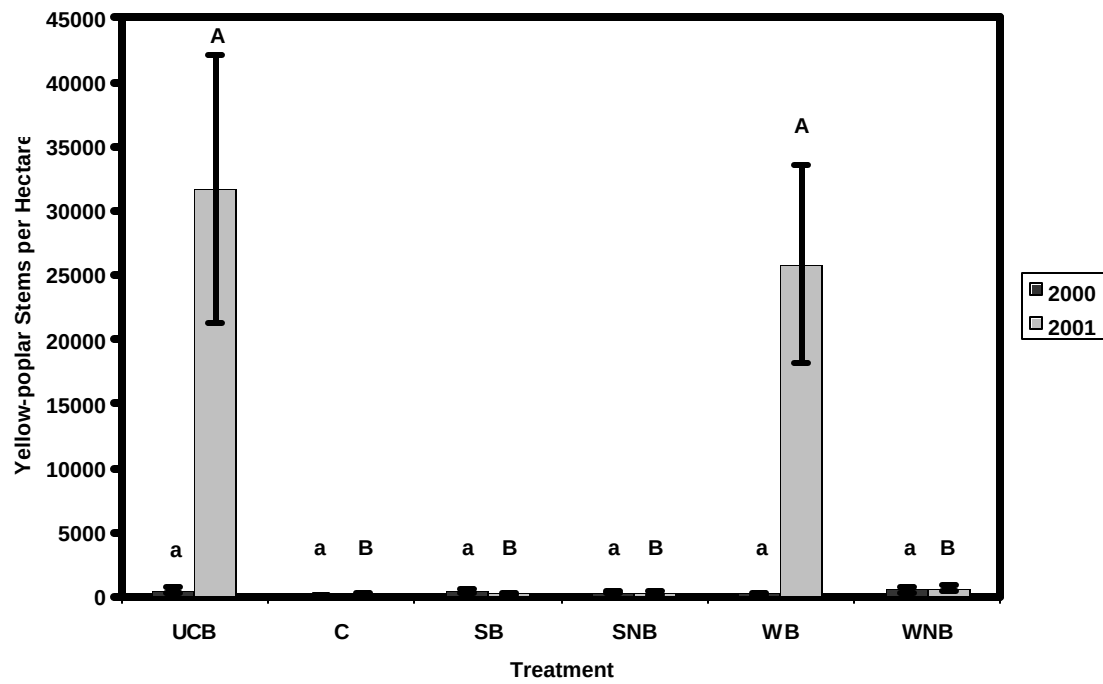


Figure 25. Yellow-poplar regeneration <10-cm tall by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

However, again the data is variable and standard errors were very large. The prescribed burning significantly increases yellow-poplar density and thus competition between oaks and yellow-poplar.

The pre-treatment total stems of regeneration per hectare <10-cm tall did not significantly differ between treatments ($P=0.6540$). Treatment means ranged from 29,115 to 38,800 stems per hectare. Post-treatment differences between treatments ($P=0.0001$) in the total stems of regeneration <10-cm tall were significant (Figure 26). The uncut burned treatment (104,207 stems/ha) and the prescribed burned wildlife thinning (121,713 stems/ha) had significantly more stems than any other treatment. Here again, the effect of prescribed burning is evident with the increase of woody regeneration <10-cm tall.

Woody Regeneration Analysis for Stems >10-cm

This section of analysis represents the woody regeneration >10-cm tall but less than 1.4-m tall. Several key species, including oaks and their primary competitors were analyzed individually and a total number of stems of woody regeneration per hectare were analyzed. Data recorded represent the number of stems per hectare of a particular species or group. Data in this section was analyzed by year to show significant differences. Table 4 also lists the species recorded in this size class.

Pre-treatment American beech stems per hectare >10-cm tall did not significantly differ between treatments ($P=0.7824$). Treatment means ranged from 512 to 1,832 American beech stems per hectare. Similar results were found for year 2001, post-treatment. American beech stems per hectare >10-cm tall did not significantly differ between treatments ($P=0.0773$). Treatment means ranged from 72 to 1710 American

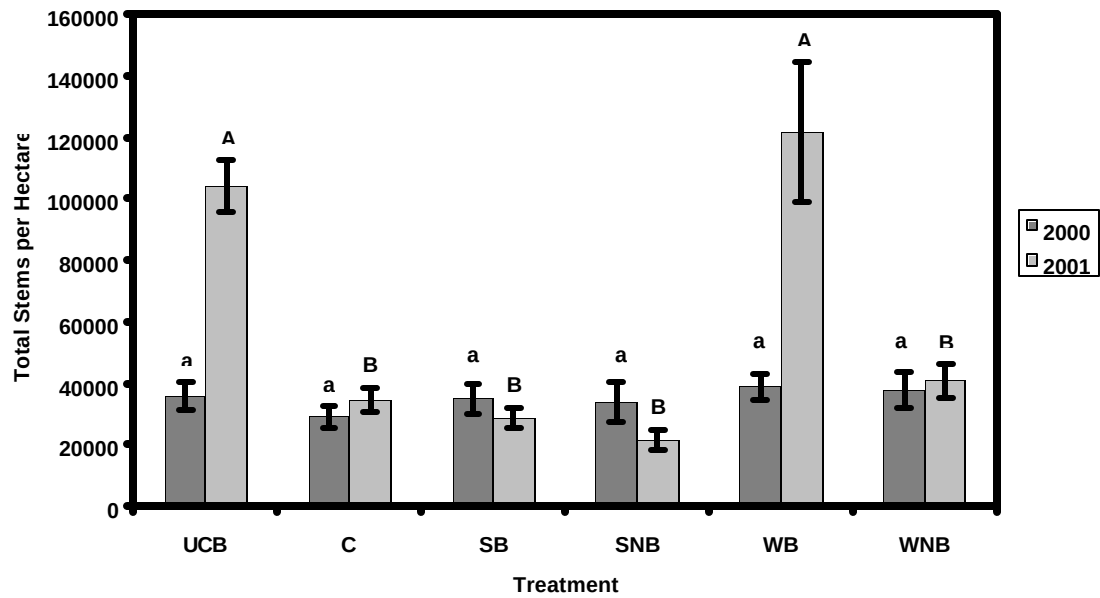


Figure 26. Total stems of regeneration <10-cm tall by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

beech stems per hectare. The natural log of this variable was used due to non-normality in the raw data.

Pre-treatment black cherry stems per hectare >10-cm tall did not significantly differ between treatments ($P=0.1838$). Treatment means ranged from 573 to 1,177 black cherry stems per hectare. Black cherry stems per hectare in post-treatment year 2001 did not significantly differ between treatments ($P=0.0813$). Treatment means ranged from 235 to 890 black cherry stems per hectare.

No pre-treatment significant treatment differences ($P=0.5459$) were found for blackgum stems per hectare >10-cm tall. Treatment means ranged from 1,054 to 2,221 blackgum stems per hectare. Significant post-treatment differences between treatments ($P=0.0281$) were detected in blackgum stems per hectare >10-cm tall (Figure 27). The prescribed burned wildlife thinning (4,515 stems/ha) had significantly more blackgum stems than the control (2,201 stems/ha) or the unburned shelterwood (1,873 stems/ha). The square root of this data was used in the analysis due to non-normality in the raw data.

Pre-treatment blueberry stems per hectare >10-cm tall did not significantly differ between treatments ($P=0.8866$). Treatment means ranged from 22,123 to 32,248 blueberry stems per hectare. No significant post-treatment differences between treatments ($P=0.8892$) were found in blueberry stems per hectare >10-cm tall. Treatment means ranged from 3,189 to 10,667 blueberry stems per hectare (Figure 28). The natural log of this variable was used due to the non-normality of the raw data. The means, though not significantly different, showed the most reduction in blueberry stems per hectare in the shelterwood cut treatments. Site and soil disturbance may be the cause of this decline.

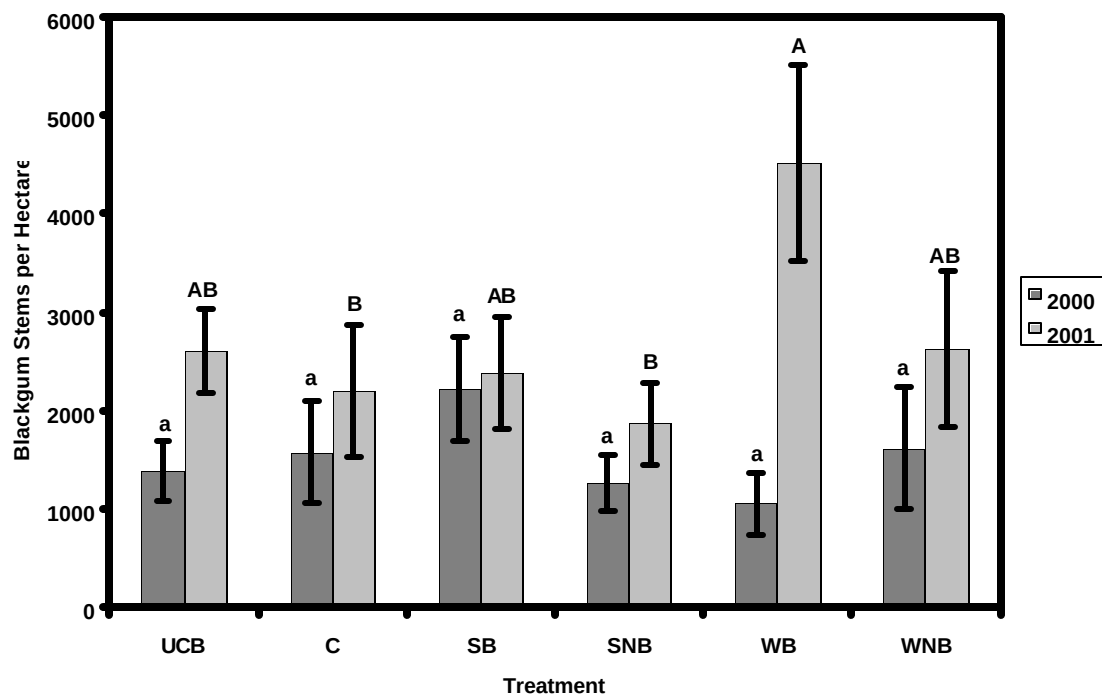


Figure 27. Blackgum regeneration >10-cm tall by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

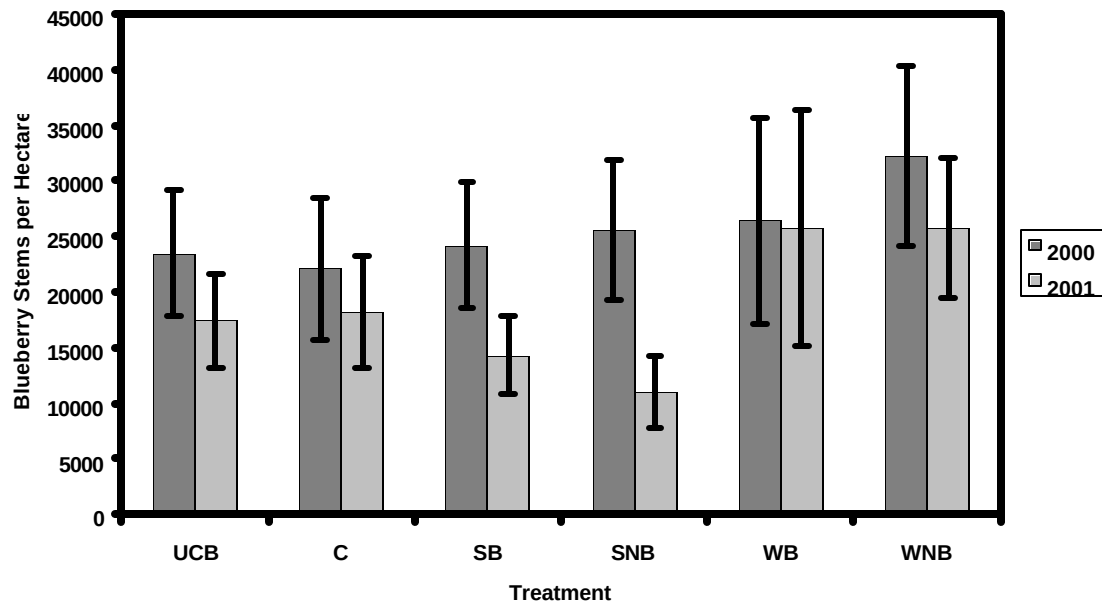


Figure 28. Blueberry regeneration >10-cm tall by treatment. Error bars and significances in Figure 3. Treatment codes as in Figure 1.

Treatment differences were not significant ($P=0.1360$) in pre-treatment chestnut oak stems per hectare >10-cm tall. Treatment means ranged from 7,361 to 19,431 chestnut oak stems per hectare. Post-treatment data indicated no significant differences between treatments ($P=0.1011$) in chestnut oak stems per hectare >10-cm tall (Figure 29). However, the prescribed burned treatments and shelterwood thinning showed the most significant reductions in chestnut oak stems from the pre-treatment data. The natural log of this variable count was used due to the non-normality of the raw data.

Pre-treatment flowering dogwood stems per hectare >10-cm tall did not significantly differ between treatments ($P=0.5779$). Treatment means ranged from 174 to 542 flowering dogwood stems per hectare. No significant differences were detected between treatments in post-treatment flowering dogwood stems per hectare >10-cm tall. Treatment means ranged from 420 to 1,740 flowering dogwood stems per hectare. The natural log of this variable count was used the non-normality of the raw data.

Pre-treatment hickory stems per hectare >10-cm tall did not significantly differ between treatments ($P=0.6389$). Treatment means ranged from 962 to 1,464 hickory stems per hectare. No significant treatment differences ($P=0.5788$) were found for post-treatment hickory stems per hectare >10-cm tall. Treatment means ranged from 583 to 1,147 hickory stems per hectare.

Treatments did not significantly differ ($P=0.9889$) in pre-treatment red maple stems per hectare >10-cm tall. Treatment means ranged from 32,443 to 39,087 red maple stems per hectare. Post-treatment differences between treatments were not significantly different ($P=0.1769$) in the count of red maple stems per hectare >10-cm tall. Treatment means ranged from 15,428 to 31,583 red maple stems per hectare. However, in

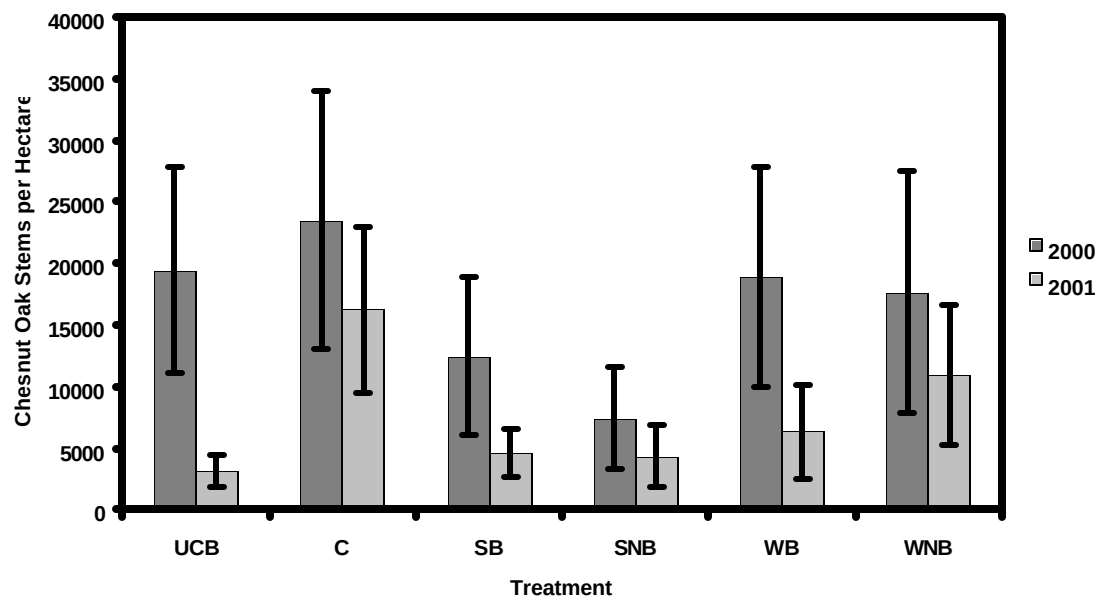


Figure 29. Chestnut oak regeneration >10-cm tall by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

comparing pre- and post-treatment levels, a reduction can be observed in the prescribed burned and disturbed treatments while not in the control in the control (Figure 30).

No significant treatment differences ($P=0.2597$) were found in pre-treatment red oak stems per hectare >10-cm tall. Treatment means ranged from 1,269 to 4,289 red oak stems per hectare. Post-treatment red oak stems per hectare >10-cm tall did not exhibit significant treatment differences ($P=0.1511$). Treatment means ranged from 983 to 3,685 red oak stems per hectare. However, the means show that in treatments that involved burning, fewer red oak stems per hectare >10-cm tall were found (Figure 31). The shelterwood treatments also exhibited some loss of red oak stems, a loss that can be attributed to site disturbance during harvesting. The natural log of this data was used due to the non-normality of the raw data.

Pre-treatment sassafras stems per hectare >10-cm tall did not significantly differ between treatments ($P=0.7592$). Treatment means ranged from 2,088 to 2,433 sassafras stems per hectare. Post-treatment treatment differences in sassafras stems per hectare >10-cm tall were marginally significant ($P=0.0553$). The mean separation did not find any significant differences between treatments and treatment means ranged from 2,866 to 21,386 sassafras stems per hectare. Sassafras >10-cm tall was most abundant in prescribed burned treatments (Figure 32). The natural log of this variable was used due to the non-normality of the raw data.

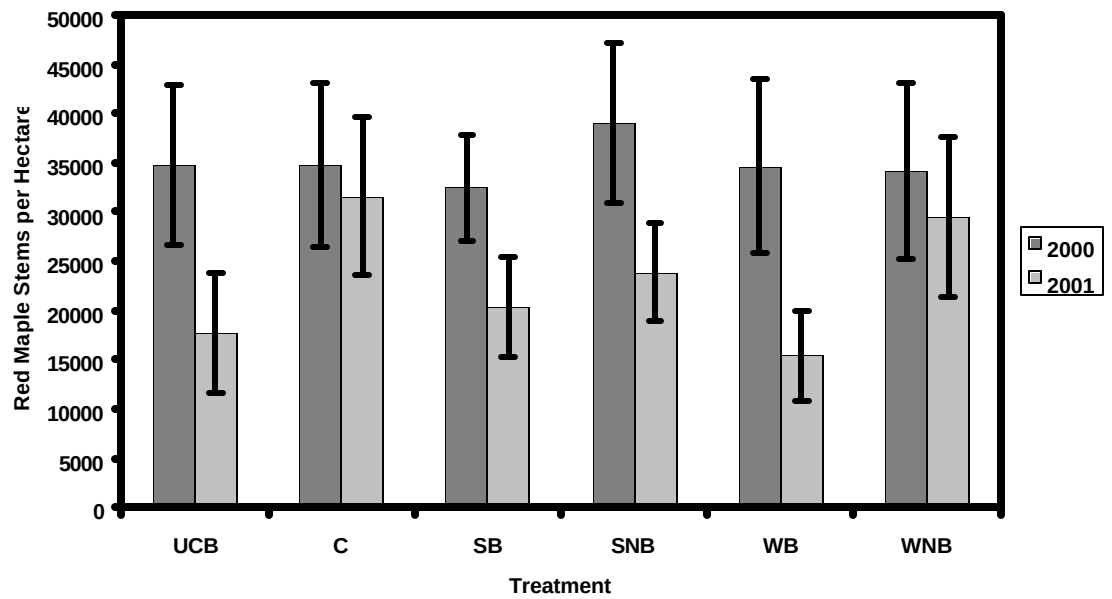


Figure 30. Red maple regeneration >10-cm tall by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

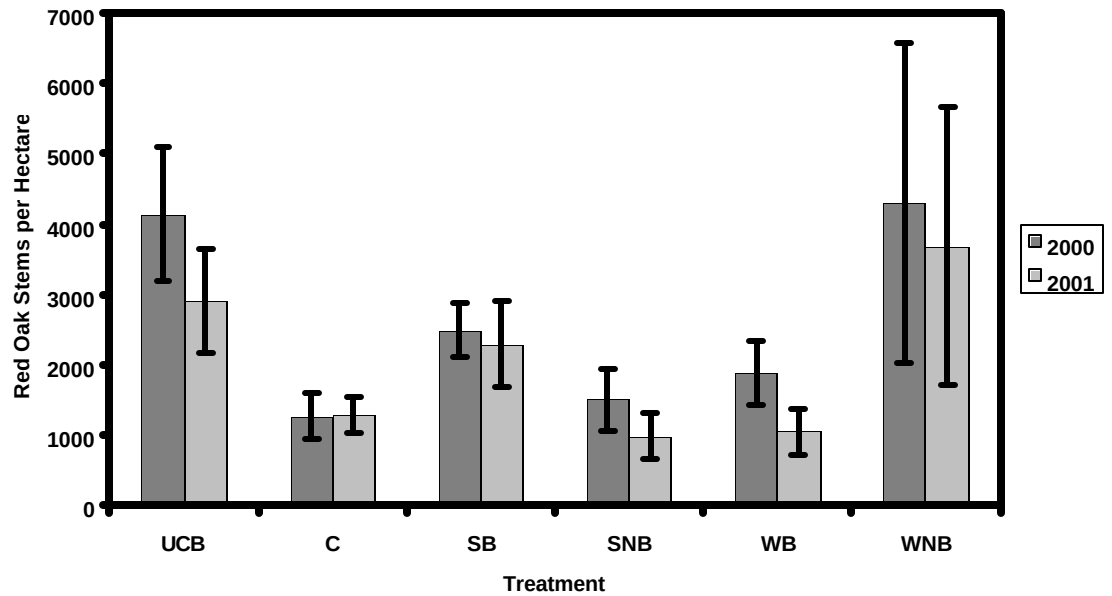


Figure 31. Red oak regeneration >10-cm tall by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

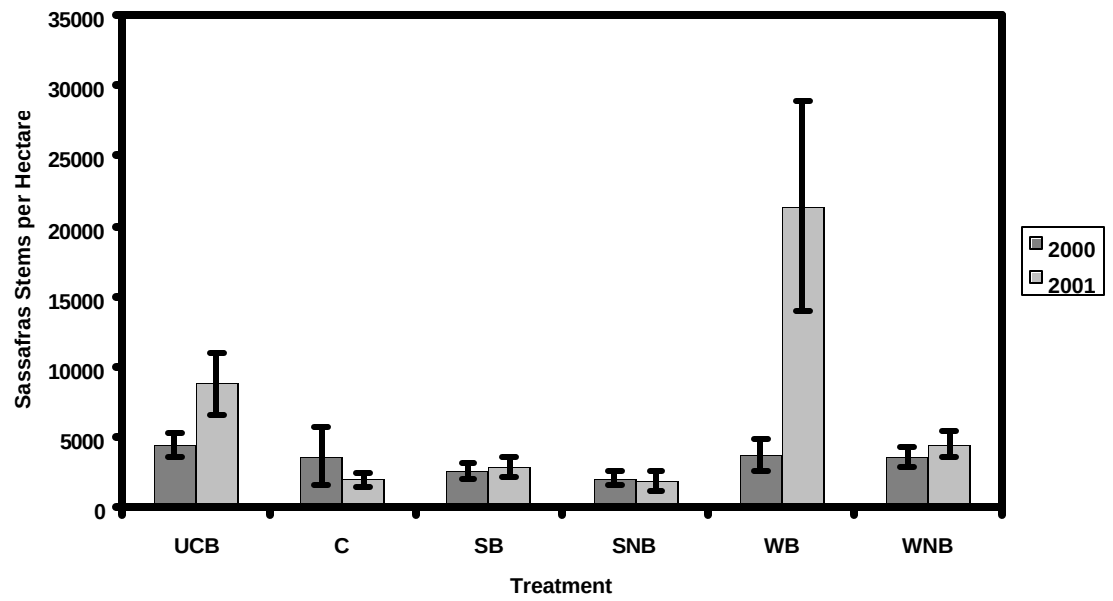


Figure 32. Sassafras regeneration >10-cm tall by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

Sugar maple stems per hectare >10-cm tall, pre-treatment, did not significantly differ between treatments ($P=0.6787$). Treatment means ranged from 491 to 2,190 sugar maple stems per hectare. Post-treatment sugar maple stems per hectare >10-cm tall did significantly differ between treatments ($P=0.0105$, Figure 33). The control (2,078 stems/ha) had significantly more sugar maple stems than the prescribed burned wildlife thinning (143 stems/ha) and the uncut burned treatment (41 stems/ha). Prescribed burning significantly reduced sugar maple. The natural log of this variable count was used due to the non-normality of the raw data.

Pre-treatment white oak stems per hectare >10-cm tall did not significantly differ between treatments ($P=0.7833$). Treatment means ranged from 1,249 to 2,887 white oak stems per hectare. Post-treatment white oak stems per hectare >10-cm tall treatment differences were marginally significant ($P=0.0529$, Figure 34). The unburned wildlife thinning (1,464 stems/ha) had significantly more stems than the prescribed burned wildlife thinning (429 stems/ha). Prescribed burning significantly reduced white oak stems in this size class. The natural log of this variable count was used due to the non-normality of the raw data.

Pre-treatment differences in yellow-poplar stems per hectare >10-cm tall were not significant ($P=0.8888$). Treatment means ranged from 491 to 1,157 yellow-poplar stems per hectare. Post-treatment yellow-poplar stems per hectare >10-cm tall differed significantly between treatments ($P=0.0279$, Figure 35). Tukey's mean separation did not find any differences between treatments. Treatment means ranged from 409 to 3,634 yellow-poplar stems per hectare. The natural log of this variable was used due to the non-normality of the raw data.

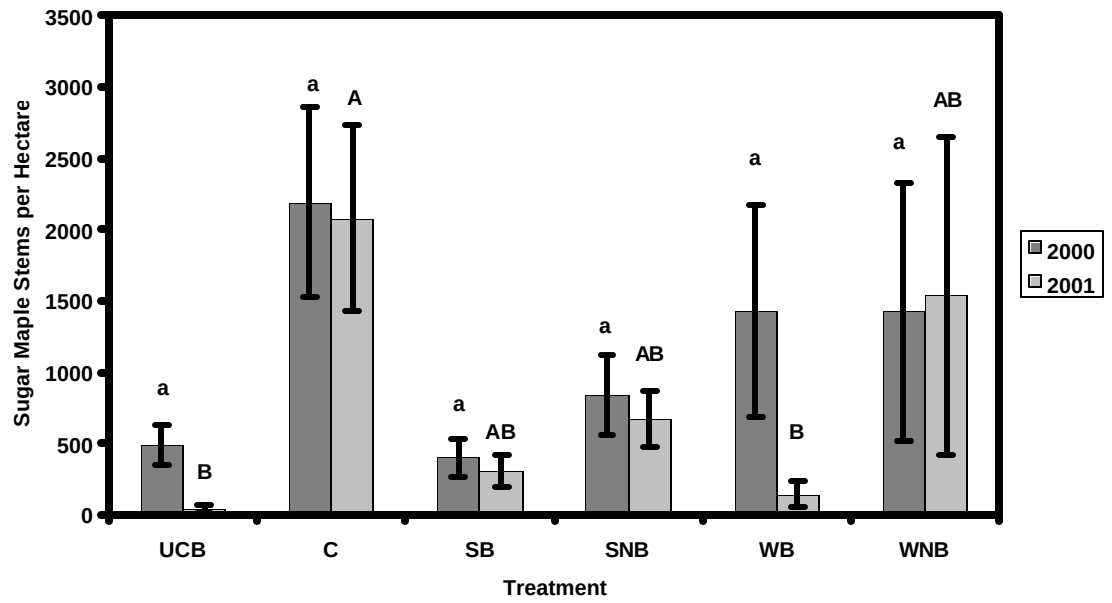


Figure 33. Sugar maple regeneration >10-cm tall by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

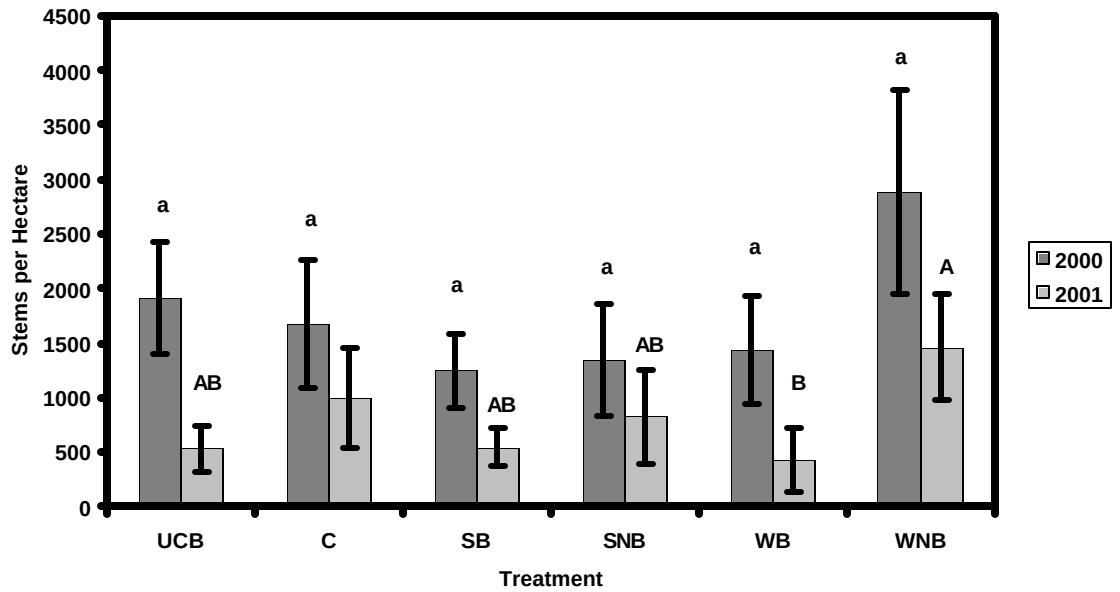


Figure 34. White oak regeneration >10-cm tall by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

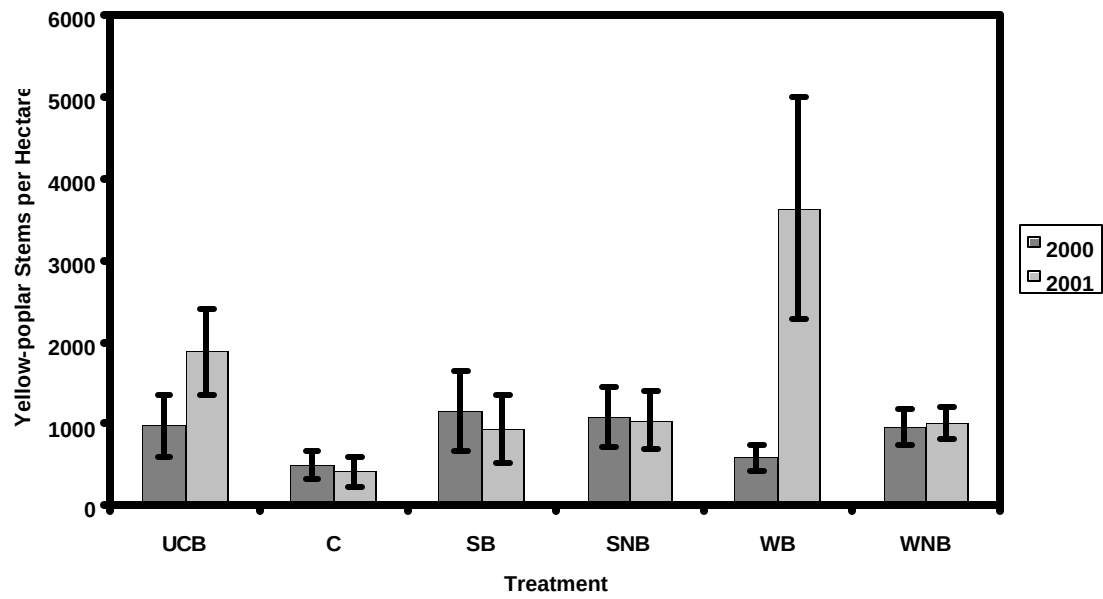


Figure 35. Yellow-poplar regeneration >10-cm tall by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

Pre-treatment total stems of woody regeneration per hectare >10-cm tall did not significantly differ between treatments ($P=0.9395$). Treatment means ranged from 84,859 to 103,522 stems per hectare. Total stems of woody regeneration per hectare >10-cm tall did not significantly differ between treatments ($P=0.3024$) in post-treatment year 2001. Treatment means ranged from 50,307 to 85,380 stems per hectare of woody regeneration (Figure 36). Shelterwood cuts showed the most dramatic reduction in stems >10-cm but less than 1.4-m tall.

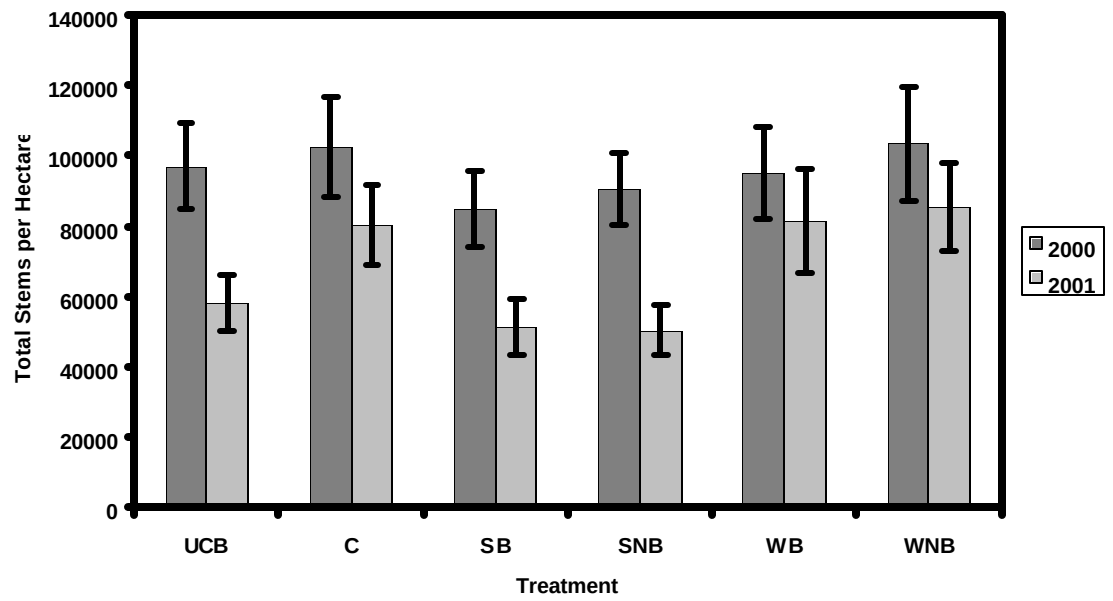


Figure 36. Total stems of regeneration >10-cm tall by treatment. Error bars and significance as in Figure 3. Treatment codes as in Figure 1.

V. DISCUSSION

Overstory Trees

The only species that exhibited overall significant differences in year 2001 between treatments was red maple. The unburned wildlife thinning had less red maple than the shelterwood cut designated to be unburned. Overstory species composition varied in density tremendously between stands. Most species were found on each site, however, in very different densities. In theory, the shelterwood cuts and wildlife thinnings should have reduced the basal area in species that are less desirable, such as red maple, sassafras, and yellow-poplar. This did not occur. If individual stands were analyzed for significant differences, we may see the expected patterns. However, drawing conclusions from a single stand on the effectiveness of the treatments would not be wise, due to site variability.

No damage or wounding to the residual overstory trees was evident in the first year following treatments in our study. However, fire scorch marks were evident on the trees. Brose and Van Lear (1999) showed that 80 percent of residual oak overstory trees had slightly damaged or altogether undamaged boles following prescribed burning. Spring fires were reported to create the highest chance of damage to residual trees. Brose and Van Lear also pointed out that shelterwood harvests leave significant amounts of slash and debris on the ground that can serve as fuel for fire. They showed that the amount of debris directly around residual trees can significantly increase the likelihood of damage. Directional felling and/or slash removal away from residual trees can reduce the likelihood of damage significantly.

Saplings

There were very few saplings on the study sites. The total number of saplings, of all species, was lowest in the wildlife thinnings, which was to be expected due to the removal of the mid-story vegetation.

With so few stems, only broad interpretations of these data are possible. The first is that oak saplings were rare to nonexistent. In year 2000, by stand, there was a maximum average of 172 white oak saplings per hectare, 36 red oak saplings, and 5 chestnut oak saplings. In year 2001, there was a maximum average of 72 white oak saplings per hectare, 14 red oak saplings, and 11 chestnut oaks. This low number of oak saplings serves as an indicator of the current oak regeneration problem. For saplings in general, wildlife thinnings tended to reduce the number of saplings in all species as did fire.

Canopy Cover

Canopy cover exhibited the expected patterns. Increased overstory removal decreased canopy cover. Prescribed burning also affected canopy cover, reducing it slightly. This is shown by the reduction in the uncut burned treatment over the control and by the increased reduction of the prescribed burned wildlife thinning over the unburned wildlife thinning. A reduction in canopy cover can be significantly correlated with the increased light availability in the understory.

Site Characteristics

The shrub crown cover responded to the treatments as expected. Any treatment involving prescribed burning had a significantly reduced shrub layer. In fact, the uncut burned treatment had no shrub cover. Arthur et al. (1998) also found that fire

significantly reduced small diameter woody stems in the midstory shrub layer. The unburned wildlife thinning also exhibited reduced shrub layer, though it still had more shrub cover than the burned areas. This can be attributed to the removal of much of the midstory in this treatment. The shelterwood cuts had a slightly reduced shrub layer due to the site disturbance of logging equipment while the control had virtually no reduction in shrub crown cover.

The basal area recorded with the prism coincided with the basal area calculated from diameter measurements of overstory trees. The shelterwood methods exhibited the most basal area reduction. We did not quite meet our goal of a residual of 11.48 square meters per hectare; however, the residual basal area in the shelterwood cuts was significantly lower than the uncut areas and the unburned wildlife thinning. The prescribed burned wildlife thinning and the unburned wildlife thinning exhibited basal areas slightly reduced from pre-treatment levels. This, too, did not meet our goal of 11.48 square meters of residual basal area per hectare. This low reduction in basal areas can be attributed to the reduced immediate kill of the herbicide. Approximately six months after herbicide application, the kill rate was only 50 percent of treated trees. Many herbicides require a lengthy time to stress and kill trees and during the next growing season, I expect there to be increased mortality in treatments where herbicide was applied. The timing of the herbicide application could also be influencing the mortality rates. Herbicides applied in early spring are generally not as effective as those applied later in the growing season. Kass and Boyette (1998) found that herbicide basal area removal was indeed less consistent than shelterwood-type cuts for overstory removal.

Light and Soil Moisture

PAR levels followed an expected pattern of increased light with increased overstory removal. With increased canopy and midstory removal, PAR levels increased in the understory. Similar increases were reported in northern Michigan (Buckley et al., 1998). The shelterwood cuts exhibited an almost 600 percent increase in PAR availability, at 1-m in height, over the control. The wildlife thinnings exhibited a 350 percent increase in PAR availability over the control. PAR measured at the 32.5-cm level showed almost identical increases. From this, we can conclude that with increased overstory removal and site disturbance, photosynthetically active radiation can be increased to give understory plants, such as advanced oak regeneration, more resources to grow. Increasing PAR availability by removing the canopy has been shown to increase the potential for seedling growth (Buckley et al., 1998). This increased growth increases the likelihood of successful regeneration establishment and recruitment into the midstory and eventually the overstory.

Soil moisture did not exhibit the expected patterns. Shelterwood cuts were expected to have increased moisture levels over the uncut areas, as were the wildlife thinning treatments. Buckley et al. (1998) reported increases in soil moisture due to canopy removal. Barnes and Van Lear (1998) showed that prescribed burning reduced soil moisture. Overall, no significant differences were found between treatments in our study. Since no differences were detected, it can be concluded that changes in soil type, elevation, and other microsite characteristics within stands could be influencing soil moisture levels more powerfully than any of our treatments. Soil type and its characteristics, such as texture, play significant roles in soil moisture (Barnes et al.,

1998). More analysis of these characteristics would be needed to better relate site characteristics to soil moisture.

Herbaceous Cover

Hardly any significant differences in herbaceous cover were found between any treatments. Herbaceous plants seemed unaffected by fire, possibly due to their quick resprouting. However, some conclusions can be drawn from this analysis. Herbaceous plants do not seem to be impeding oak regeneration at the current time. Buckley et al. (1998) concluded that the effects of herbaceous plants on oak regeneration was minor relative to overstory cover. Similarly, Beck (1970) showed that until the overstory was removed or treated, oak regeneration did not respond to understory vegetation removal. Once the effects of the shelterwood harvests and wildlife thinnings on herbs become more apparent, herbs may limit oak regeneration development and would need to be controlled, possibly through burning or another type of disturbance. Beck (1970) gives a recommended time line of understory vegetation control up to 5 years after overstory treatment.

Analysis of the species groups provides more descriptive and interpretable results. Forb coverage differed significantly by treatment. The shelterwood cuts exhibited significantly less forb coverage than the control. This can be attributed to site disturbance from logging activities. The other treatments, prescribed burning and wildlife thinning, showed reductions in forb coverages that were not significant. Burning, in the short term, may reduce forb coverages. However, increased sprouting of forbs was noticed after the burning. Once these plants mature over successive growing seasons under increased light, forb coverages may actually increase in the absence of further site disturbance. DeSelm

and Clebsch (1991) showed that forb coverage increased after prescribed burning and that periodic prescribed burning was needed to maintain this increase.

Woody vine coverage was very sparse in the treatments. Graminoids exhibited a similar pattern with little coverage in any treatment. None had graminoid coverage exceeding 0.54 percent. Woody shrubs occupied even less of the study sites. No mean woody shrub coverage over 0.003 percent was recorded for any treatment. Elliott et al. (1999) showed that both grasses and woody shrubs increased after prescribed burning, seemingly contradicting our results of no change in either group's coverage. Fern coverage, although higher, still covered little of the treatments. Mean fern coverage did not exceed 1.69 percent. Greenbrier, which exhibited no significant change between treatments, has been shown to decrease after prescribed burning (DeSelm and Clebsch, 1991). These species groups, although important for diversity, seemed unimportant in terms of overall herbaceous cover, both in terms of wildlife habitat and competitive impacts on woody regeneration.

Woody Regeneration <10-cm

Burning increased blackgum density in this size class. Blackgum significantly increased in stems per hectare in treatments that were prescribed burned. The wildlife thinning burned had a slightly higher level of blackgum than the uncut burned treatment, most likely due to the increased light availability in the wildlife thinning, where the overstory was removed.

Chestnut oak data were highly variable and produced less reliable results due to large standard errors. Statistically, there were more chestnut oak stems in the prescribed burned wildlife thinning than the shelterwood cuts. Burning seemed to increase chestnut

oaks. Stand 2 had significantly more chestnut oaks than any stand by far, but in looking at the analysis of treatments only in that stand, no clearer pattern develops. No significant differences in the red oak group or white oaks between treatments were found. However, the means indicate differences between the burned and unburned treatments. Overall, red oaks slightly increased in the uncut burned treatment when compared to pretreatment levels. The overall white oak data shows a more pronounced effect. White oak stems in prescribed burned treatments had almost 9 times the amount of stems <10-cm tall than any other treatment.

For oaks overall, Brose and Van Lear (1998) found that oak densities increased two years after prescribed burning, especially stems <0.3-m tall. This study also revealed that oak stem quality and growth rate increased in burned areas. Our study indicated significant increases in oak regeneration in the smallest size classes, some of which came from new seedling establishment. Further study over the next 2 to 3 years will be needed to document growth and other characteristics. Brose and Van Lear (1999b) also showed oaks gained a better competitive position after prescribed burning due to the reduction of competitors. Huddle and Pallardy (1999) showed that oak resprouting was significantly higher than that of red maple, thus bettering the oaks' competitive position. The increase in oak density on our study seems to indicate similar results. However, future burning may be necessary to further increase the oaks' competitive advantage over other competitors, such as yellow-poplar and sassafras, that is likely to further develop from the overstory removal.

Hickory exhibited significant differences between treatments. Though the mean separation technique did not indicate these significant differences, the means show that

prescribed burning significantly encourages hickory sprouting. Hickory follows a hypogeal sprouting pattern similar to oaks (Graney, 1990). The dormant buds on the root collar are below the soil surface, thus protected by fire. This protection enables hickory to readily resprout after disturbance, such as fire. Elliot et al. (1999) showed that hickory species, along with oaks, were the most abundant woody regeneration species following burning.

Burning encouraged two of oaks' most abundant competitors in the <10-cm size class. Both sassafras and yellow poplar responded rapidly to fire. Prescribed burned treatments supported significantly more sassafras stems less than 10-cm tall than any other treatment. The uncut burned treatment had almost 900 percent more sassafras than the closest unburned treatment, the shelterwood designated to be burned. The two burned treatments had had at least 84 times more yellow-poplar than any other treatment. Brose and Van Lear (1998) found that yellow-poplar exhibited a growth rate increase for up to one year after burning, only to return to pre-burn rates. Their study sites contained significant quantities of yellow-poplar regeneration prior to burning, due to the burning being conducted under 2-4 year old shelterwood harvests. Barnes and Van Lear (1998) reported that up to 71 percent of existing yellow poplar could be reduced using prescribed burning. Our study sites had little yellow-poplar and sassafras regeneration prior to the treatments being installed. The prolific response of these species may indicate a need to conduct further prescribed burning on the sites to get the reduction in stems of competitive species that these studies accomplished.

Red maple was the oaks' most abundant competitor in the pretreatment year. When comparing the two years, any treatment involving site disturbance or prescribed

burning produced reductions in red maple density, while the control and unburned wildlife thinning saw increases in red maple. Brose and Van Lear (1998) found significant reductions in red maple following spring prescribed fires. Red maple seedlings fire survival was reported to be at least half that of oaks of similar size in Missouri (Huddle and Pallardy, 1999).

In terms of total stems of regeneration, of all species, treatments that involved prescribed burning showed significantly more stems <10-cm tall than other treatments. Elliott et al. (1999) showed that the total number of seedlings significantly increased after burning.

For woody regeneration under 10-cm tall, prescribed burning increased the number of oak stems, as well as the number of competitor stems, such as sassafras and yellow-poplar. Prescribed burning substantially reduced red maple of this size class.

Woody Regeneration >10-cm

Only three separate species exhibited significant differences by treatment. Blackgum and yellow-poplar were both encouraged by prescribed burning while sugar maple was reduced by prescribed burning. Treatments involving prescribed burning had significantly more stems of yellow-poplar than any other treatments. Results for yellow-poplar and sassafras in the >10-cm tall size class were very similar to those discussed for the <10-cm tall size class. Sassafras was more abundant in prescribed burned treatments, significantly increasing over pre-treatment levels. The other primary competitor of oak did not show clear differences. When comparing pre- and post-treatment levels of red maple, a reduction can be observed in the prescribed burned and disturbed treatments while not in the control in the control. Overall, red maple >10-cm but less than 1.4-m tall

seems to be susceptible to prescribed fire and site disturbance. This is consistent with the results of other studies (Elliott et al., 1999).

Chestnut oak exhibited no first-year response to the treatments. This could be attributed to several things. It was expected to find less chestnut oak >10-cm in the burned areas due to top kill by the fire. However, chestnut oak sprouted quickly after the fire and many stems may have reached over 10cm in height following the fire. There was a reduction in stems per hectare in taller sizes. In treatments that involved burning, fewer red oak stems per hectare >10-cm tall were found. The shelterwood treatments also exhibited some loss of red oak stems, a loss that can be attributed to site disturbance during harvesting. The fire top-killed the red oak but the data shows that it readily sprouted back into the smaller size class. White oak treatment differences occurred in a similar manner to those of red oaks. However, larger numbers of white oak entered the <10-cm size class after burning than were top-killed in the larger size class. Burned and disturbed areas had fewer white oaks of this size class. Little research has been conducted on the effects of prescribed fire on oaks of this particular size class. It is widely held, however, that stems of this size, in any species, will be significantly reduced by fire and site disturbance.

VI. SUMMARY AND CONCLUSIONS

The first objective of the research was to evaluate and compare the effectiveness of the shelterwood harvests and wildlife thinnings in terms of their ability to stimulate oak advanced reproduction. Both of these treatments significantly increased understory light availability by reducing canopy cover. Light at both the 100-cm and 32.5-cm levels increased when any overstory thinning was conducted. The wildlife thinning treatments increased light availability relative to the uncut areas, but not as substantially as the shelterwood cutting. The prescribed burned wildlife thinning treatment had slightly greater PAR levels than the unburned wildlife thinning. Overstory thinning combined with prescribed burning maximized understory light availability. This increased light availability likely benefits oak regeneration. First-year responses in the oaks are somewhat difficult to document due to the short growing time since the treatments were installed. Increased light availability, however, has been shown through many studies to stimulate oak regeneration (e.g. Beck, 1970; Buckley et al., 1998).

The effectiveness of the prescribed burning in stimulating oaks and enhancing the effects of the overstory removal was also documented, especially on white oaks. Prescribed burning significantly affected oaks. Oaks in the large seedling class, those over 10-cm tall but less than 1.4-m tall, are significantly reduced due to the top kill of vegetation by the fire. Stems in the smaller size class, those less than 10-cm tall, were significantly increased with burning. This increase in oak stems can be attributed to two sources: new seedling establishment (white oaks) and new stems of white oak and all other oaks sprouting from the root stock of the existing oak regeneration that was top killed in the fire. Fire also increased PAR in the understory by reducing the shrub layer.

Both overstory removal treatments and prescribed burning had mixed results on herbaceous, shrub, and non-oak tree species. The herbaceous plant community was virtually unaffected in the first year following treatments. Fire introduced new species as did site disturbance from logging, but neither treatment significantly altered the herb richness of the sites. Fireweed was introduced by fire and pokeberry by site disturbance. Fire seemed to encourage the growth of herbaceous plants, especially forbs. It is expected that, over time, an enhanced herbaceous layer will develop in the areas that were prescribed burned. Both the wildlife thinnings and the prescribed burning significantly reduced the shrub layer, thus increasing growing space and other resources for oaks. Shelterwood harvests, through site disturbance, reduced the shrub layer somewhat, but not significantly. Prescribed burning likely increased competition between oaks and yellow-poplar and sassafras. Competition from red maple was reduced in all regeneration size classes. The increase in yellow-poplar and sassafras likely occurred due to seed already in the soil prior to the treatments, and sprouting. Prescribed burning, combined with overstory removal, provided the seedbed and increase in light necessary for those seeds to sprout. I believe subsequent fires may be needed on a fairly frequent interval to reduce the number of yellow-poplar and sassafras stems. Provided there is a sufficient fuel load, burning can be carried out on intervals as little as every 2 to 3 years to reduce competition while not harming the oak stems (David Van Lear, Personal Communication). Subsequent fires should eventually reduce the ratio of these competitors to oaks.

Overstory removal significantly increased PAR availability. Burning further reduced the midstory structure to allow even more light to reach the seedling level. None

of the treatments significantly affected measured soil moisture levels. It is possible that site characteristics (e.g. drainage patterns and soil type) not measured in this study played a more important role in regulating soil moisture than the treatments.

Based on effects on PAR and certain competitors, such as red maple, shelterwood cuts, wildlife thinnings, and prescribed fire may benefit oak regeneration on the study sites. Shelterwoods provide economic return to the landowner while wildlife thinnings reduce understory and soil disturbance. If the shelterwoods on this study were conducted by a landowner, they would have provided an income of \$28,845 and a volume of almost 86,000 board feet over the approximately 13 hectares harvested. Further income would be realized when the residual trees were removed. Using herbicides for wildlife thinnings represents an investment for the landowner. Kochenderfer et al. (2001) determined that using Garlon 3A in injection treatments of trees cost about \$11.19 per square meter of basal area treated for removal. This amounts to a cost of approximately \$128 per hectare to conduct the thinnings on the study sites, on which 11.5 square meters per hectare of basal area was removed. In addition to reducing certain competitors, this type of overstory and midstory reduction can benefit wildlife habitat by reducing site disturbance while increasing light availability to the forest floor. Prescribed burns benefit both wildlife and timber production interests. Fire is an economical way to encourage herbaceous understories and control hardwood competition to release the preferred oaks. Studies have shown prescribed burning to cost an average of \$36.20 per hectare for all aspects of burning combined. The cost of this management practice can be justified when considering the long-term benefits of ensuring continued oak dominance in the overstory

and the continued quality of wildlife habitat. It is up to the individual landowner to decide what treatments are most appropriate to meet their goals.

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