# BIENNIAL SEASONAL BURNING AND HARDWOOD CONTROL EFFECTS ON THE CARBON SEQUESTRATION IN A NATURAL LONGLEAF PINE ECOSYSTEM

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# BIENNIAL SEASONAL BURNING AND HARDWOOD CONTROL EFFECTS ON THE CARBON SEQUESTRATION IN A NATURAL LONGLEAF PINE ECOSYSTEM

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BIENNIAL SEASONAL BURNING AND HARDWOOD CONTROL

EFFECTS ON THE CARBON SEQUESTRATION IN A NATURAL

LONGLEAF PINE ECOSYSTEM

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# THESIS ABSTRACT

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This study has been superimposed on a study established in 1973 on the Escambia Experimental Forest located in south-central Alabama, USA to examine the effects of different seasons of burn and hardwood control treatments on longleaf pine overstory growth and understory plant succession. The study aims to examine the relationship between various seasons of prescribed fire (winter, spring, summer, and no-burn) and supplemental hardwood control treatments (one-time chemical, periodic mechanical, and untreated check) on carbon sequestration in natural longleaf pine stands. Overstory longleaf pine trees were measured and understory vegetation and litter samples were collected in September 2003 to determine biomass and percent carbon. Soils were sampled at three depths, 0-10, 10-20 and below 20 cm, to determine percent carbon.

Analysis of variance was run to test the effects of treatments on carbon content in the understory vegetation and mineral soil. Average DBH and height of longleaf pine trees were greater on no-burn plots. No significant effects of burning and supplemental hardwood treatments on the basal area and biomass of the longleaf trees at stand level were observed. Significantly higher total biomass carbon was documented in the no burn plots, but the total biomass carbon did not differ significantly among burning treatments. The effect of biennial burning on carbon content was primarily limited to the upper 0.1 m of the mineral soil with little change apparent in the depth below 0.1 m. No burn plots had the highest carbon stored in the soil and summer burn plots had the highest carbon content among the burning treatments in 2006. No burn plots had the highest carbon stored in the soil for chemical and control plots of supplemental hardwood treatments in 2007. An increase in soil carbon was observed in the upper 0.1 m layer of mineral soil during one year time period however there was decrease in carbon in depth below 0.1 m. No burn plots had highest amount of carbon stored in the soil in year 2006 and 2007. However, the increase was lowest in these plots with spring burn plots having the highest increase in soil carbon in upper 10 cm layer during this one year time period.

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# 1. INTRODUCTION

Global atmospheric carbon dioxide ($CO_2$) concentration level is increasing steadily over preindustrial level due to various anthropogenic activities particularly burning fossil fuels (Bolin, 1970; Baes et al., 1977) and depletion of forests and biomass burning (Bolin, 1977; Schneider, 1989). The anthropogenic input to the atmosphere has almost tripled over the past five decades. Atmospheric $CO_2$ levels are increasing at the rate of 0.4 percent per year and are estimated to double during the 21$^{st}$ century. Under a scenario of a growing global economy and without controls on emissions, the atmospheric concentration of $CO_2$ is expected to rise to 700 ppm (parts per million) or more from current 380 ppm (IPCC, 2001).

Atmospheric carbon dioxide, being one of the major greenhouse gases, contributes about 63 percent of the gaseous radiative forcing responsible for anthropogenic climate change (Hofmann et al., 2006) thereby trapping exiting solar radiation from the earth. The increase in anthropogenic greenhouse gases, particularly atmospheric $CO_2$, is more likely to result in an increase in global atmospheric and oceanic temperatures. The increase in globally averaged temperatures induces changes in global climate system. The issue of carbon sequestration has gained momentum globally after the questions of "carbon budgets" and "carbon credits" moved to the forefront in the post-Kyoto era. A potential mechanism for reducing the net carbon emission into the atmosphere is through increased biological sequestration in biomass and soil in forest ecosystems (Jackson and Schlesinger, 2004).

Longleaf pine (*Pinus palustris* Mill.) ecosystems once occupied an estimated 25-35 million hectares (Frost, 1993). The natural range of longleaf pine includes most of the Atlantic and Gulf Coastal Plains from southeastern Virginia to eastern Texas, and into central Florida and the Piedmont and mountains of north Alabama and northwest Georgia (Boyer, 1990; Stout and Marion, 1993). Longleaf pine occurs in range of sites from wet, poorly drained flatwoods near the coast to mesic uplands, xeric sandhills, and dry, rocky mountain ridges (Boyer, 1990). This ecosystem is distinguished by an open, park-like stand structure "pine barren" which typically comprises even-aged and multi-aged mosaics of forests, woodlands, and savannas with bunch grasses (wiregrass and certain bluestems) dominated diverse groundcover, and understory free from hardwoods and brush (Landers et al., 1995). Pine barrens are known for their significant persistence and diversity and their ecological persistence is attributed to a product of long-term interactions among climate, fire, and traits of the key plant species (Landers et al., 1995).

Longleaf pine ecosystems are maintained by fires and might eventually disappear altogether from a site in the event of fire absence for a long period, particularly on fertile sites with aggressive hardwood competition (Hermann, 1995). Natural fires occurred every 2 to 8 years in the longleaf pine range prior to landscape fragmentation (Christensen, 1981; Abrahamson and Harnett, 1990). Frequent low intensity fire, occurring every 2 to 4 years, helps to meet almost all of the biological requirements for natural regeneration of this species (Landers et al., 1995; Boyer, 1999). In absence of fire plant communities of less fire-adapted species encroach and replace the longleaf communities (Ware et al., 1993; Engstrom et al., 2001).

Since the colonial times the longleaf pine ecosystem has been intensively exploited due to its many desirable attributes (Croker, 1979). The cumulative impacts of changing land uses over the last three centuries by European settlers and Native Americans resulted in dramatic decline of longleaf forest in its natural range. One of the major historical factors responsible for the final disappearance of the longleaf pine after initial exploitation for agriculture, logging, and naval stores was the implementation of fire suppression policy during the 1920s (Frost, 2006). However, recognizing the ecological role of fire in longleaf pine ecosystems, the U.S. Forest Service was using prescribed fire as fuel reduction tool by the 1940s (Pyne et al., 1996). Only a small portion of the historic range of longleaf pine ecosystem (about 3.2 million ha) are prescribe-burned in the entire southern US (Wade et al., 2000). The rapidly expanding wildland-urban interface is a growing challenge that might prove daunting to implement prescribed fire in the long run.

Longleaf pine ecosystem is one of the forest ecosystems that could be utilized to increase terrestrial carbon sequestration in the southeastern U.S. This study examines the relationship between prescribed burning and above ground biomass and carbon sequestration in a natural longleaf pine forest. It assesses the potential for soil carbon sequestration in natural longleaf pine stands.

## 2. OBJECTIVES

A study initiated in 1973 in south-central Alabama, U.S.A. was used to determine the effects of hardwood control treatments on understory plant succession and overstory growth in natural stands of longleaf pine (Boyer, 1983). Boyer (1995) reported on responses of understory vegetation before, seven, and nine years after treatments. Kush et al. (1999, 2000) examined effects of 23 years of these treatments on the long-term response of understory vegetation in naturally regenerated longleaf pine forests. Using the study initiated by Boyer, this study aims to examine effects of fire on carbon sequestration in longleaf pine ecosystem. The specific objectives of this study are:

1. Determine the effects of prescribed burning treatments combined with supplemental hardwood control treatments on above ground biomass and carbon sequestration.
2. Determine the effects of prescribed burning on soil carbon sequestration in natural longleaf pine stands.

# 3. LITERATURE REVIEW

## 3.1. Longleaf Pine Ecosystem

During the Wisconsinan Ice Age (about 40,000 to 12,000 years before present), forests in the southern regions were profoundly influenced by boreal species (*Pinus*, *Picea*) and temperate species (*Carya*, *Castanea*, *Ostrya*, *Quercus*) intermixed in a pattern that varied spatially and temporally with the retreat of the glaciers further to the north (Watts, 1970; Delcourt, 1980; Watts et al., 1992). As the glaciers retreated, the climate experienced periodic warming and cooling, vegetation patterns in the South changed rapidly, and species moved northward and westward from their Ice Age refuges. Following the continental glacier retreat, southern forests became dominated by oaks and various deciduous hardwoods after 12,000 year BP (Watts, 1971; Watts and Hansen, 1988; Watts et al., 1992). After the Quaternary Period, climate in the southeast changed and was characterized by the increased summer warmth, moisture, storm activity, drought severity, and lightning frequency (Delcourt et al., 1993). The longleaf ecosystem became dominant in the lower Coastal Plain ~ 8,000 years ago (Watts et al., 1992) and expanded its range throughout the Southeast during the following 4,000 years (Delcourt and Delcourt, 1987). Although climate, soil, and topography shape vegetation distribution, the dominant ecological process that influenced and maintained the composition structure, and function of the longleaf pine ecosystem was frequent burning (Landers et al., 1995).

The long-term result of frequent surface fires is a forest ecosystem that contains primarily pyrophytic vegetation and biological diversity that is adapted to the prevailing disturbance regime (Landers et al., 1995; Brockway and Lewis, 1997; Engstrom et al., 2001). The pattern, structure, and biodiversity in these forest ecosystems are sustained by the combination of disturbance and site factors. Variability in the disturbance regime is brought in by lightning strikes, treefalls, and animals influences at the local levels and tropical storms, soil properties, and hydrological extremes influence the landscape mosaic (Landers et al., 1995). Several attributes of longleaf pine forests, such as sustained population of federally endangered and threatened wildlife populations, wiregrass (*Aristida stricta*) dominated ground cover, and undisturbed upland-wetland ecotones, reflect the diversity and ecological functionality of this ecosystem.

The longleaf pine ecosystem supports a substantial proportion of the biological diversity of the southeastern Unites States that includes a high percentage of species with endangered status (Noss, 1988; Plat, 1999; Means, 2006). The total number of resident vertebrates, including specialists or endemics, is greater than for any other habitat type in the Coastal Plain of the southeastern United States (Means, 2006). The longleaf pine ecosystem is the most species-rich vegetation communities outside of the tropics (Peet and Allard, 1993) and is one of the most species-rich terrestrial ecosystems in the temperate United States (Wahlenberg, 1946; Hardin and White, 1989). The structure and function of fire-maintained longleaf pine forests maintain high levels of biological diversity and heterogeneity in the understory (Brockway and Outcalt, 1998). The vascular species richness of longleaf pine ecosystem has been recorded near 40 species/$m^2$ (Walker and Peet, 1983) or 140 species in 1000 $m^2$ area (Peet and Allard, 1993). Grasses, such as wiregrass (*Schizachyrium scoparium* and *Aristida* spp.) and bluestems (*Andropogon* spp.), that dominate the ground cover in longleaf pine savannas facilitate fire that is required to maintain the longleaf pine ecosystem structure and composition (Frost et al., 1986; Noss, 1989). Frequent surface fire encourages high level of herbaceous species diversity in longleaf pine ground cover (Battaglia et al., 2003).

### 3.2. The Role of Fire in Longleaf Pine Ecosystem

The role of disturbance (natural or anthropogenic) in maintaining and sustaining many ecosystems is extensively recognized (White, 1979). Frequent low intensity surface fire is an essential component of the longleaf pine ecosystem (Landers et al., 1995; Brockway and Lewis, 1997; Outcalt, 2000). Longleaf pine and bunch grasses ensemble functions as keystone species that facilitate the ignition and spread of surface fire during humid growing seasons (Landers, 1991). Continuous wiregrass-ground cover dominated by *Aristida stricta* (wiregrass) in longleaf pine stands suggests a history of frequent fire and the history lacking root disturbance (agricultural activities). These are critical conservation attributes from a landscape perspective (Kirkman and Mitchell, 2006). Frequent fire was mostly responsible for the competitive success of longleaf pine and the grasses. These species have distinct fire tolerance, longevity, and nutrient and water retention capacity that reinforce their site dominance and curb plant community change ensuing disturbance (Landers et al., 1995).

Historically, the dominant sources of ignition included lightning, Native Americans, and European settlers. Lightning fires have prevailed for many years (Robbins and Myers, 1992) in the southeastern U.S. and is the primary selective force promoting the development of fire-adapted traits in plant and animal communities of those regions. Lightning fires are more common during the growing season (Komarek, 1974; Noss, 1989) and prevent species native to other habitats from encroaching into longleaf habitats (Landers et al., 1995). The Native Americans used fire as their primary tool for the management of the landscape for their benefit such as reducing fuels and protecting themselves from wildfires, enhancing wildlife habitat and their population, aid in hunting, preparing land for agriculture, and reducing insects (Bonnichsen et al., 1987; Williams, 1989; Bonnicksen, 2000; Carroll et al., 2002). Combination of long history of Native American-induced and lightning-caused fire helped genetically fix fire-tolerance characteristics in species in the longleaf ecosystem (Masters et al., 2003). Landers and Wade (1994) hypothesized that the longleaf ecosystem persists because the interaction of climate-site-fire-plant emphasizes the dominance of the longleaf pine-bunchgrass ensemble. The chronic fire regime also maintained the soil nutrient dynamics and soil morphology to which longleaf pine is more adapted (McKee, 1982). European settlers also made extensive use of fire (Komarek, 1974) and expanded the use and frequency of fire throughout the South blending their indigenous fire knowledge with that of Native Americans (Wade et al., 2000). They implemented the practice of periodically burning nearby woodlands and forests to improve forage quality for cattle and prevent understory hardwood encroachment. Although longleaf was well adapted to frequent surface fire, it was not well adopted to other disturbances brought by those settlers.

Longleaf pine is self-perpetuating species that produces large pyrogenic needles that facilitate fire (Grace and Platt, 1995). In the longleaf pine-bunchgrass ensemble, fine tinder provided by bunch grass (*Aristida* spp., *Andropogon* spp., *Sorghastrum* spp., *Schizachyrium* spp., and others) and dead, resinous needles of longleaf pine furnishes fuel that ignites readily and spreads quickly across the open landscape (Clewell, 1989; Noss, 1989; Landers, 1991). Dominance of longleaf pine over large areas is primarily attributed to its comparative tolerance to frequent fire over competing species with thinner-barked seedlings. Low-intensity fires rarely kill overstory longleaf pine individuals due to their thick fire-resistant bark (Myers, 1990). The roots, bole, and crown of longleaf possess characteristics that make this species comparatively fire resistant compared to other southern pine species. Its exceptional adaptation to its fire-prone environment is a juvenile “grass stage” that favors root growth and tufts of long needles concentrated at seedling top which surround and protect a large terminal bud (Brockway et al., 2005; Moser and Wade, 2005). It has thick root collar which stores enough food reserves along with the tap root and helps the grass stage longleaf seedling to grow 1 to 2 cm during the first year putting the terminal bud beyond the lethal reach of most surface fires. Fires help in natural pruning thereby creating a clear bole between the crown and surface fuels. The clear bole coupled with the natural propensity of longleaf pine to regenerate more successfully in open forests reduce ladder fuels near the crown of mature trees (Grace and Platt, 1995; Brockway et al., 2005). Thick bark of longleaf pine protects vascular cambium from the lethal heat from surface fires (Wahlenberg, 1946). Sixty-nine percent of the mammal species and over thirty-three percent of bird species associated with longleaf ecosystem forage primarily on or near the forest floor, highlighting the ecological role of surface fire in maintaining ground cover for mammalian and avian communities (Engstrom, 1993).

### 3.3. Atmospheric Carbon Dioxide and Carbon Sequestration

About 10 petagrams of carbon (Pg C) are released to the atmosphere worldwide in the form of $CO_2$ annually by fossil fuel burning and deforestation with more than half captured by the oceans and the terrestrial biosphere (Baker, 2007). The United States has the highest rate of annual anthropogenic carbon emission in the world. The estimated global fossil-fuel $CO_2$ emission in 2004 was about 7.9 Pg C and the U.S. share of fossil fuel-related $CO_2$ emission for the same year was 1.7 Pg C (Marland et al., 2006), approximately 22 percent of the world's total. Carbon dioxide is the primary greenhouse gas released as a result of anthropogenic activities in the U.S., representing approximately 83.9 percent of total greenhouse gas emissions (EPA, 2007). Fossil fuel combustion alone accounted for 94 percent of total $CO_2$ emissions in 2005 for the United States and overall U.S. total emissions have increased by 16.3 percent from 1990 to 2005 (EPA, 2007). There has been increased international pressure to reduce the net carbon emission around the world and the United States itself. The United Nations Framework Convention on Climate Change, signed by various developing and developed countries including United States at the June 1992 Earth Summit, required member countries to develop national emission limits and emission inventories and limit emissions of $CO_2$ to

1990 levels by 2000 (Parson et al., 1992). Terrestrial ecosystems act as a major sink for carbon by removing carbon dioxide from the atmosphere through photosynthesis and storing in the plant biomass. Over the course of time, some part of the biomass is converted into stable soil carbon.

Globally, forest ecosystems cover more than 4.1 billion hectares of the Earth's land surface (Dixon and Wisniewski, 1995) and account for about 70 percent of the carbon exchange between the atmosphere and land (Schlesinger, 1997). The global forests have been estimated to hold nearly 80 percent of all above-ground terrestrial carbon and about 40 percent of all below-ground (soils, litter, and roots) carbon (Dixon et al., 1994). Globally, forest vegetation and soils contain a total of about 1150 Pg of C with two-thirds of the terrestrial carbon in forest ecosystems contained in soils (Dixon et al., 1994). The terrestrial ecosystems in North America have the largest terrestrial uptake of global terrestrial ecosystems and are the major carbon sink (Fan et al., 1998, Pacala et al., 2001). Fan et al., (1998) suggested that 1.7 ± 0.5 Pg C year$^{-1}$ is taken up in the terrestrial ecosystems of North America, in contrast to 0.1 Pg C year$^{-1}$ taken up in Eurasia. A terrestrial sink of this magnitude could utterly offset North America's emissions from fossil fuel of 1.6 Pg C year$^{-1}$ (Fan et al., 1998). The current biological sequestration of forest ecosystems in the United States is about 0.2 Pg C year$^{-1}$ from the atmosphere (Heath and Smith, 2004), a carbon sink equivalent to about 10 percent of U.S. emissions of $CO_2$ from fossil fuels combustion. Houghton et al. (1999) estimated an average accumulation of 0.037 Pg C year$^{-1}$ for the 1980s in the United States due to land-use change and the net flux of carbon attributable to management of terrestrial ecosystems offsets 10 to 30 percent of U.S. fossil fuel emissions. Pacala et al. (2001) estimated a carbon sink in the conterminous United States between 0.30 and 0.58 Pg C year$^{-1}$ on an average during the decades of the 1980s. Similarly, Heath et al. (2001) estimated that U.S. forest ecosystems currently remove 0.2 – 0.3 Pg C year$^{-1}$ from the atmosphere as terrestrial carbon in the form of biota and in soil. In one study, Birdsey and Heath (1997) estimated that US forests have sequestered enough carbon over the past four decades to offset approximately 25 percent of $CO_2$ emission in the U.S.

The forest ecosystems of the Southeastern USA have been functioning as carbon sink since the 1950s (Delcourt and Harris, 1980) and the southern U.S. has the strongest biological carbon sink in the U.S. (Turner et al., 1995). Schimel et al. (2000) estimated the net carbon storage in the terrestrial ecosystems of the U.S. as 0.08 Pg C year$^{-1}$ and the annual net carbon storage per unit area for the Southeast (150 kg ha$^{-1}$) was the highest among different bioclimatic regions of the U.S. The southern United States represents approximately 25 percent of the land area, 60 percent of the forest land, and 25 percent of the agricultural land of the entire United States and hence, southern forests play an extensive role in the terrestrial sequestration of atmospheric carbon, accounting for approximately 29 percent of the aboveground carbon stock in the conterminous United States (Mickler, 2004). Both managed and natural southern pine forests have played a major role in offsetting the atmospheric $CO_2$ emission. These forests sequester or store carbon in both *in situ* pool (in vegetation biomass and soils) and *ex situ* pool (in the form of final products) thus may help the United States to meet national carbon emissions commitment (Johnsen et al., 2001). The terrestrial ecosystems also store a large amount of carbon in soils for the longest period of time (Schlesinger, 1997). Since the forest ecosystems occupy large areas at regional and global scale, forest soils play a crucial role in the global carbon cycle (Detwiler and Hall, 1988; Sedjo, 1992; Bouwman and Leemans, 1995). The terrestrial humus has been estimated to contain as much as one to four times the amount of carbon in the living biota of forest ecosystems (Bohn, 1976; Baes et al., 1977). The soil carbon pool contains as much as 85 percent of the terrestrial carbon in the high-latitude boreal forests, 60 percent in the mid-latitude temperate forests, and 50 percent in low-latitude tropical forests (Dixon et al., 1994). Schimel (1995) estimated that globally the carbon stock in the soils (1580 Pg C) is approximately twice as much as in the atmosphere (750 Pg C) or terrestrial vegetation (610 Pg C). Forest soil has tendency to store more carbon than soil under agriculture (Guggenberger et al., 1994) and thus, plays crucial role for the global carbon cycle. Forest soils contain about 60 percent of the total terrestrial ecosystem carbon in the United States (Birdsey and Heath, 1995). Atmospheric $CO_2$ is stored in the forest soil in stable solid form by two primary mechanisms; first by direct fixation which involves soil carbon sequestration as inorganic soil carbon compounds through inorganic chemical reactions, and second by indirect fixation that involves soil organic carbons sequestered through decomposition of plant biomass (Kimble et al., 2001). Soil and forest sinks could be used by the U.S. to meet half of its carbon emission reduction commitment (Eve et al., 2000).

### 3.4. Ecological and Biological Characteristics of Longleaf Pine That Make it Superior to Other Southern Pine Species for Carbon Sequestration

Longleaf pine is the longest-living of southern pines species and has been reported to have a maximum biological potential to reach or exceed an age of 450 – 500 years (Platt et al., 1988; Landers et al., 1995). However, the species is unlikely to survive to this biological potential due to the exposure of these forests to frequent disturbances such as lightning or wind (Palik and Pederson, 1996). It continues to grow and respond to release even at older ages (West et al., 1993). Since longleaf pine outlives other southern pine species and continues to put on growth even at older ages, longleaf pine is likely to sequester carbon for longer time periods than other pines. West et al. (1993) suggested an increase in annual increments of all age classes in old-growth longleaf pine trees of age ranging from 100 to nearly 400 years. He reported an average annual ring increment in 1987 was 40 percent greater than in 1950 and the increase in annual increments for 100 to 150-years old trees was approximately 45 percent and for 200 to 400-years old trees was 35 percent when compared with expected annual increment.

Unlike other southern pines, it tolerates wide range of habitats ranging from wet, poorly drained flatwoods near the coast to excessively drained sandhills and dry, rocky mountain ridges (Boyer, 1990). It grows naturally on nutrients poor soils and the natural altitudinal distribution of this species ranges from sea level along the Atlantic and Gulf Coastal Plains to a height of 580 m above mean sea level in east central Alabama (Boyer, 1990). Longleaf pine adaptability to a range of site conditions and longevity makes this species more suited for carbon sequestration in the southeast United States as compared to other important hardwood species.

Longleaf pine is less susceptible to various damaging agents as compared to other southern pine species. It has a natural resistance to fire and the more serious pathogenic agents that afflict other southern pines, including fusiform rust (*Cronartium* spp.), annosus root rot (*Heterobasidion annosum*), pitch canker (*Fusarium moniliforme* var. *subglutinans*), southern pine beetle (*Dendroctonus frontalis* Zimmermann), and several species of coleopteran bark beetle (*Dendroctonus* and *lps* spp.) (Wahlenberg, 1946; Anderson and Doggett, 1993; Walkinshaw et al., 1993; Landers et al., 1995). Longleaf pine is also more resistant than slash pine (*Pinus elliottii*) to ice storm breakage (Van Lear and Saucier, 1973) and has resistance to windthrow and uprooting from hurricanes due to its massive tap root that may reach a depth of 2.5 to 3.5 m in mature trees (Wahlengerg, 1946; Boyer, 1990). It has a competitive advantage over other southern pines and hardwoods in area with frequent surface fires. Loblolly pine (*P. taeda*) and slash pines are thinner barked and often susceptible to fire-caused mortality but the longleaf pine, due to its thicker bark, is adapted to fire (Wahlengerg, 1946). Fire, which is an integral part of longleaf ecosystems, helps control brown-spot needle blight (*Scirrhia acicola*) that can severely limit the growth and survival of longleaf seedlings (Boyer, 1975).

Since specific gravity or density of the wood is an important indicator of quality of wood products, product utilization, and long-term decay rates, it is an important factor for any tree species from the carbon storage capacity perspective. Based on a large sample of age classes and site locations, the specific gravity of longleaf pine (based on green volume and oven dried weight) was found to be 8 to 12 percent higher than other southern pines on an average (Koch, 1972) and hence, produces more dry weight per unit of volume. The average specific gravity of longleaf pine was 0.57 as compared to 0.53 in slash pine, 0.51 in loblolly pine, and 0.52 in shortleaf pine (*P. echinata*). Zobel et al. (1972) reported that longleaf pine produced wood with a higher specific gravity than both slash and loblolly pine when grown under same site conditions. Comparatively higher specific gravity of longleaf pine gives this species competitive advantage over other southern pine species for carbon storage for longer time period. The superior growth form and wood quality makes this species more suited for long-term *ex-situ* carbon storage pool in the form of final products over other southern pines. Typically about 50 to 80 percent of the trees in naturally regenerated longleaf forests produce high-quality poles, pilings, log, post, and peelers for plywood (Boyer and White, 1990; Landers et al., 1995). The durability of final products produced from longleaf pine trees accentuates the potential of this species as a source of long-term carbon storage.

Fire-maintained longleaf pine ecosystems are the most species-rich plant communities outside of the tropics (Peet and Allard, 1993) and support a productive understory of a great variety of herbaceous plant species. A mesic longleaf pine ecosystem has been reported to contain up to 140 vascular plant species per 1000 $m^2$ area (Peet and Allard, 1993). This productive understory of diverse plant species also offers an additional opportunity for carbon sequestration in addition to the longleaf pine tree itself. An approach for increasing terrestrial carbon in longleaf pine ecosystems not only offers biological sequestration in the southern United States but also helps to save the second most endangered ecosystem in the U.S. (Noss et al., 1995) and provide habitats to federally endangered and threatened flora and fauna associated with this ecosystem.

### 3.5. Prescribed Fire and Carbon Sequestration

The gross natural $CO_2$ emissions by forest fires, detritus decomposition, and plant respiration is offset by photosynthetic uptake into organic matter as the annual increment of growth of the terrestrial biota and tropical vegetation on land recovering ensuing shifting cultivation rotation (Wong, 1978). Wong (1978) estimated that the total non-fossil fuel burning input from forest fires and land-clearing fire release 5.7 Pg C year $^{-1}$ into the atmosphere with the gross carbon input due to the temperate and boreal forest burning at 0.47 Pg C year $^{-1}$. However, Fahnestock (1979) claimed that Wong's estimates of non-fossil fuel burning input are overestimated and hence the net carbon input from the forest burning is much less significant. Fahnestock (1979) estimated a gross carbon input for the same temperate and boreal areas burned annually to be only 0.11 Pg C year $^{-1}$ as compared to Wong's 0.47 Pg C year $^{-1}$ with the gross carbon input from the prescribed burning of forest debris in the temperate zone for management purposes as $\leq$ 0.02 Pg C year $^{-1}$ which is not significant enough to contribute to the atmospheric carbon dioxide level increase. Thus the carbon dioxide contribution from the prescribed burning does not appreciably affect the atmospheric $CO_2$ budget.

Fire is one of the major disturbances that impact soil carbon dynamics in forest ecosystems (Wells et al., 1979; Lal, 2005). Its impact on soil organic carbon stock depends on the temperature and duration of fire, amount of soil organic carbon and its distribution in the soil profile, and change in the soil organic carbon decomposition rate following the fire event (Page-Dumroese et al., 2003). Forest wildfires result in greater losses of soil carbon than most prescribed fires (Johnson, 1992) and prescribed fire reduces intensity, size and damage from wildfire by reducing and removing the portion of accumulated dead and live fuel loads (Fernandes and Botelho, 2003; Liu, 2004). Johnson (1992) reported that the losses of soil carbon from mineral soil due to low-intensity prescribed fire were insignificant or nonexistent as compared to wildfires. It has been hypothesized that increased growth rates following low-intensity prescribed fires compensate for the carbon emissions during biomass burnings, resulting in a negligible net effect on atmospheric and ecosystem carbon budget (Crutzen and Goldammer, 1993) and the nitrogen losses due to burning is also compensated from increased $N_2$ fixation by legumes following burning (Wells, 1971; Waldrop et al., 1987; Boring et al., 1991). In some cases, there was a marked increase in soil carbon following the prescribed burning as a result of establishment of post-fire nitrogen-fixing plant species (Johnson and Curtis, 2001). Binkley et al. (1992) examined the 30-year cumulative effects of prescribed fires at intervals of 1, 2, 3, and 4 year on soil chemistry in loblolly/longleaf pine forest in the Coastal Plain of South Carolina. Surface carbon content per unit area in mineral soils (0-20 cm depth) was higher in the treatment plots than in the control (fires suppressed) plots, although there was no trend related to burning interval.

Prescribed burning may also reduce $CO_2$ emission along with other greenhouse gas emissions from wildfires. Narayan et al. (2007) estimated the emissions from wildfires in the European region over a 5-year period to be approximately 11 million tonnes of $CO_2$ per year and with the application of prescribed burning the emission dropped down to about 6 million tonnes, a potential reduction of almost 50 percent. Similarly, in their study Narayan et al. (2007) discussed a summary of the results of the study by Fernandes (2005) in maritime pine stands in Portugal. The results indicated that the release of $CO_2$ and other compounds from long-term prescribed burning was about 62 percent lower than the emissions from a more severe wildfire.

# 4. METHODS

## 4.1. Study Area

This study was conducted on the Escambia Experimental Forest (EEF) located in Escambia County, Alabama, U.S.A. at $31^0$ 01' N mean latitude and $87^0$ 04' W mean longitude. The 1200 hectares EEF was established in 1947 when the T.R. Miller Company of Brewton, Alabama provided the land at no cost to the USDA Forest Service through a 99 year lease (Boyer et al., 1997).

The climate is humid and mild with abundant rainfall well distributed throughout the year. July and August are the warmest months with average daily maximum and minimum temperatures of $33^0$ C and $20^0$ C, respectively. December and January are the coldest months with average daily temperatures of $18^0$ C and $3^0$ C, respectively. Average annual precipitation is about 156 centimeters and October is the driest month. The predominant soil series on this coastal plain site is the Troup series with Wagram and Dothan soils also present. The Troup series is very low in natural fertility with low organic matter content. The soil in this area is formed in unconsolidated marine sediments of loamy sands, sandy loams, and sandy clay loams. The available water capacity is low or very low in the sandy layers and medium in the loamy layers.

### 4.2. Experimental Design and Treatments

This study was superimposed on a study initiated in 1973 to examine the effects of hardwood control treatments and fire on growth of dominant pine overstory and effects on understory vegetation succession in natural stands of longleaf pine (Boyer, 1983). Boyer (1983) described the establishment, methods, and treatment regimes for this study. The relatively uniform and even-aged stands were established from the 1958 seed crop and were released from the parent overstory during the winter of 1961. The study was a Randomized Complete Block Design (RCBD) with two types of treatments randomly assigned. Four burning treatments included biennial prescribed burns in winter (December to February), spring (April, May), summer (July, August), and a no-burn check. Each burning treatment was combined with three supplemental hardwood treatments. These were: (1) an initial treatment of hardwoods and woody shrubs injected with metered amounts (1 ml per 2.54 cm diameter at breast height) of undiluted 2,4-D amine during the late spring of 1973 (woody stems too small to inject were wounded or cut with the injector bit, and the metered amount of herbicide was allowed to flow over the wound); (2) a periodic mechanical treatment (hand-clearing of all hardwood stems greater than 1.3 m in height in 1973 and at regular intervals thereafter, as needed); and (3) an untreated check that received no supplemental hardwood treatment. All treatments were replicated in three blocks. Each block consisted of 12 square, 0.16 hectare treatment plots with a 0.04 ha measurement plot nested in each 0.16 ha treatment plot. The twelve treatment combinations were randomly assigned among the 12 treatment plots in each of three blocks.

In 1973, when the study was installed, all the plots were thinned to 1250 pines per ha, yielding 50 pines in each 0.04 ha measurement plot. The longleaf pine stands were 14 years old from seed and 12 years from parent overstory removal. They had an average height of 6.7 m and DBH of 8.1 cm.

### 4.3. Vegetation Sampling

The longleaf pine overstory was measured for diameter and total height in early September 2003. All hardwood trees with a DBH greater than 1 cm were recorded for their DBH and total height. All the living material less than 1 cm DBH was destructively sampled from nine systematically located 0.89 $m^2$ sample plots per treatment plot in late September/ early October 2003. The vegetation was sorted by species based on taxonomy of several authorities (Grelen and Duvall, 1966; Radford et al., 1968; Clewell, 1985; Godfrey, 1988; Kartesz, 1994) and into four different vegetation components: grasses, forbs, woody vines, and shrubs. Litter was sampled from one 30.5 $cm^2$ subplot within each of the 0.89 $m^2$ sample plots. The vegetation and litter was oven-dried at $70^0$ C for 72 hours and weighed. For each component, a sub-sample of the dried material was ground with a Wiley Mill and sieved to be used for percent carbon determination.

### 4.4. Soil Sampling

The existing guidelines for soil carbon accounting refer only to the upper 0.3 m for soil sampling which is intended to cover the actively changing soil carbon pool (IPCC, 1997). Soil samples were collected using a stainless steel probe from each of the nine 0.89 $m^2$ sample plots per treatment plot in early May 2006 and 2007. Soils cores were taken at three different depths from the surface: 0-0.1 m, 0.1-0.2 m, and from 0.2 m to a maximum depth of 0.35 m. The later depth varied as in some cases the soil was too hard to probe to 0.35 m. The slopes were minimal in all the plots.

### 4.5. Basal Area and Biomass of Longleaf Pine Trees

Basal area of longleaf pine trees is defined as the cross sectional area of a tree measured at breast height (DBH) or 1.37 m and was calculated as

BA ($m^2$) = 0.00007854 $DBH^2$ (in cm) and reported as $m^2ha^{-1}$

Aboveground biomass of longleaf pine trees (including needles) were predicted using a regression equation of natural longleaf pine trees developed by Taras and Clark III (1977). The best independent variables examined for estimation of aboveground biomass of longleaf pine trees were DBH and total tree height with the equation

$$Y = \beta_0 + \beta_1 D^2Th + \varepsilon \quad \textbf{Eq. 4.5.1}$$

Where,

Y = biomass of a tree (pounds),

D = DBH in inches,

Th = total tree height in feet,

$\varepsilon$ = experimental error, and

$\beta_0, \beta_1$ = regression coefficients

Taras and Clark III (1977) took the logarithm of Y to stabilize the variance of Y that increased with increasing $D^2Th$. Final regression equation developed for estimating aboveground biomass of longleaf pine trees (including needles) was

$$\text{Log}_{10}\, Y = -0.99717 + 1.00242\ \text{Log}_{10}\, D^2Th \quad \ldots\ldots\ldots\ldots\ldots\ldots \textbf{Eq. 4.5.2}$$

with coefficient of determination ($R^2$) = 0.99 and standard error = 0.036.

Although tree biomass was calculated in English unit (pound), later it was expressed in metric unit (kg).

### 4.6. Carbon Analyses Protocol

Carbon in the understory vegetation and soil was determined quantitatively using a Thermo Finnigan Flash 1112 N/C analyzer. Processed samples were run on the analyzer according to the machine's standard operating instructions. Twenty percent of all the samples were duplicated to check the instrument's repeatability. The accuracy of the sample values was checked using one NBS standard and one CE Elantech Inc. certified standard in each sample set. A sample set consisted of 31 samples, two certified standards, a blank (empty tin capsule), and six random duplicate samples. Coefficients of variations (CV) for each duplicate sample were generated after the samples had been run. The sample was rerun if the CV was higher than five percent. This continued until the CV was lower than five percent. Entire sample sets were reweighed and rerun if standards were not within ten percent of certified standard values.

The amount of carbon content in each of the understory components (grasses, forbs, litter, woody vines, and shrubs) in the longleaf pine stand was calculated by multiplying the dry-weight of the samples by carbon percentage of the samples.

### 4.7. Statistical Procedures

SAS (Statistical Analysis System) statistical software version 9.1 (SAS Institute Inc., 2003) was used for most statistical data analyses. Statistical tests were conducted at 5% level of significance.

Two-factors ANOVA was performed to examine the main effects of supplemental hardwood treatments and burning treatments on DBH, height, basal area, and biomass of longleaf pine trees. PROC GLM of SAS was used to conduct the test. Type III sums of squares were presented for ANOVA effects because unbalanced data were used in the analyses. The results of the different tests were compared before final conclusions were determined. When the interaction effect was significant Tukey-Kramer test was used to perform multiple comparisons.

Two-factors ANOVA was performed to test main effects of the supplemental hardwood treatments and burning treatments on dry-weights and carbon content of non-longleaf pine understory components with understory components (grasses, forbs, litters, woody vines, and shrubs) as blocking variable in the test. The statistical model used for the two-factor blocked factorial design with 3 replications was

$$y_{ijk} = \mu + \tau_i + \beta_j + (\tau\beta)_{ij} + \delta_k + \epsilon_{ijk} \quad \text{…………………… } \mathbf{Eq.\ 4.7.1}$$

Where,

$\mu$ = overall mean

$\tau_i$ = the effect of the $i^{th}$ level of supplemental hardwood treatments

$\beta_j$ = the effect of the $j^{th}$ level of burning treatments

$\delta_k$ = the effect of the $k^{th}$ block i.e. five different understory vegetation components

$(\tau\beta)_{ij}$ = the effect of the interaction between the $i^{th}$ level of supplemental hardwood treatments and the $j^{th}$ level of burning treatments

$\epsilon_{ijk}$ = the random error

$i$ = 1, 2, 3 (three different supplemental hardwood treatments, i.e. chemical, control, and mechanical)

$j$ = 1, 2, 3, 4 (four different burning treatments, i.e., no burn, winter, spring, and summer)

$k$ = 1, 2, 3, 4, 5 (five different understory components i.e. grasses, forbs, litters, woody vines, and shrubs)

PROC GLM and Tukey-Kramer test were used to perform multiple comparisons. Similarly the main effects of supplemental hardwood treatments and burning treatments on carbon contents in understory components were tested using the same model as for dry-weight.

The main effects of supplemental hardwood treatments and burning treatments on soil carbon in longleaf pine were tested using PROC GLM in the two-factor ANOVA with three soil sample depths (i.e. 0-0.1 m, 0.1-0.2 m, and below 0.2 m) as blocking variable. The statistical model used for this analysis was

$$y_{ijk} = \mu + \tau_i + \beta_j + (\tau\beta)_{ij} + \delta_k + \epsilon_{ijk} \quad \text{……………………} \ \textbf{Eq. 4.7.2}$$

where, all the symbols denote the same variables as in Eq. 4.7.1 except for $k$ which in this equation denotes three different soil depths i.e. 0-0.1 m, 0.1-0.2 m, and 0.2 m below the soil surface and thus, has values from 1 to 3. Multiple comparisons of main effects were performed using Tukey-Kramer test and Dunnett's *t* test was used to compare main effects of control treatment with the main effects of other treatments.

An additional analysis of soil carbon was done considering time and depth as factors. Four-way repeated measurement designs with repeated measures on time was performed to examine the effects of hardwood treatments and burning treatments on soil carbon contents at three different depths in 2006 and 2007. Mauchly's Sphericity test was not necessary to do for this date set. Since the measurement on the subject was made twice, compound symmetry is automatically satisfied which eliminates the need to perform the Sphericity test (Huynh and Feldt, 1970). The statistical model used for this design was

$$y_{ijkl} = \mu + \beta_i + \gamma_j + \tau_k + (\beta\gamma)_{ij} + (\beta\tau)_{ik} + (\gamma\tau)_{jk} + (\beta\gamma\tau)_{ijk} + S_l + (\beta S)_{il} + (\gamma S)_{jl} + (\tau S)_{kl} + (\beta\gamma S)_{ijl} + (\beta\tau S)_{ikl} + (\gamma\tau S)_{jkl} + (\beta\gamma\tau S)_{ijkl} + \epsilon_{ijkl} \quad \text{...... } \textbf{Eq. 4.7.3}$$

where,

the fixed effects were:

$\beta_i$, $\gamma_j$, and $\tau_k$ are the main effects of supplemental hardwood treatments, burning treatments, and time period (2006 and 2007) respectively

$(\beta\gamma)_{ij}$, $(\beta\tau)_{ik}$, $(\gamma\tau)_{jk}$, and $(\beta\gamma\tau)_{ijk}$, are the interaction effects of supplemental hardwood treatments and burning treatments, supplemental hardwood treatments and time, burning treatments and time, and supplemental hardwood treatments, burning treatments, and time respectively

$S_l \sim N(0, \sigma_1^2)$ is the subject *l* effect i.e. three different soil depths

$\epsilon_{ijkl} \sim N(0, \sigma^2)$

Analyses were conducted to check any potential violation of basic assumptions of ANOVA. Normality of residuals was checked using both graphical tests (a normal probability plot or a normal Q-Q plot) and formal tests (Anderson-Darling and Cramer-von Mises based goodness-of-fit tests for normal distribution). PROC UNIVARIATE was used to determine normality of the residuals. Suitable data transformations were applied if the assumption of normality was not satisfied. Similarly, the homogeneity of variance was checked using both the graphical and formal tests. Residual plots (plots obtained when the residuals are plotted against the predicted values) provided useful information about the homogeneity of variance. Both Levene's test and Brown and Forsythe's test for homogeneity were used to check assumption of constant variance. Suitable transformations were used where applicable to fix non-constant variance in the data.

# 5. RESULTS AND DISCUSSION

## 5.1. Treatments and Stand Characteristics

ANOVA analyses showed that the supplemental hardwood treatments had no effects on DBH (Table 5.1.1) and height (Table 5.1.2) of the longleaf stand whereas the burning treatments had significant effects (Table 5.1.1 and Table 5.1.2). The basic statistical measures of DBH and height of the longleaf pine trees encountered in the sample plots are given in Appendix 5.1.1.

Table 5.1.1. Testing the effects of supplemental hardwood and burning treatments on DBH (cm) of longleaf pine trees

| **Source** | **DF** | **Type III SS** | **Mean Square** | **F-value** | **Pr > F** |
|---|---|---|---|---|---|
| Supplemental | 2 | 142.50 | 71.25 | 3.03 | 0.0500 |
| Burning | 3 | 1030.89 | 343.63 | 14.60 | < .0001 |
| Supplemental*Burning | 6 | 1225.76 | 204.29 | 8.68 | < .0001 |

Table 5.1.2. Testing the effects of supplemental hardwood and burning treatments on height (m) of longleaf pine trees

| **Source** | **DF** | **Type III SS** | **Mean Square** | **F-value** | **Pr > F** |
|---|---|---|---|---|---|
| Supplemental | 2 | 9.68 | 4.84 | 0.67 | 0.5457 |
| Burning | 3 | 304.87 | 101.62 | 12.72 | < .0001 |
| Supplemental*Burning | 6 | 407.81 | 67.97 | 8.51 | < .0001 |

However, since the interaction between the supplemental hardwood treatments and burning treatments was significant for DBH (Table 5.1.1) and height (Table 5.1.2), and hence the effects of burning treatments on the longleaf pine height were examined at each level of supplemental hardwood treatment. Average DBH (Fig 5.1.1) was significantly greater on unburned plots than on winter and summer burned plots in chemical treatments, greater on unburned than on winter burned plots in mechanical treatments, and greater on unburned, winter, and spring burned plots than on summer burned plots in control or no supplemental hardwood treatments (Appendix 5.1.2). Boyer (1993) reported similar results for the longleaf pine stands characteristics in the same experimental forests examined in 1989. However, he did not note any significant interaction between the supplemental hardwood treatments and burning treatments.

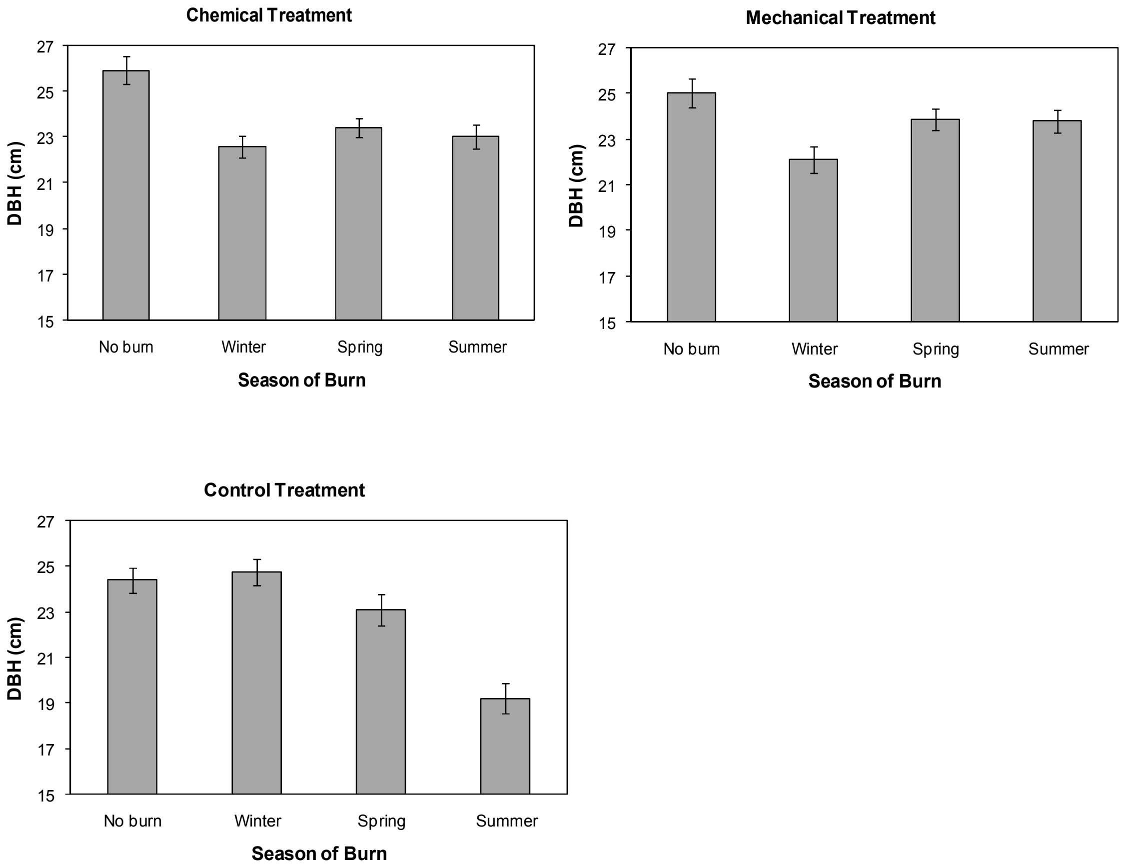


Fig 5.1.1. Average longleaf pine DBH (cm) by burning treatment at different level of supplemental hardwood treatments. Bars represent standard errors of the means

The average height was not significantly different among no burn and burning treatments in chemical supplemental treatment (Fig 5.1.2). In mechanically treated plots no burn treatments had the highest height and significantly higher than the winter burn treatments (Appendix 5.1.2). Differences among season of burn were not significant. In control supplemental hardwood treatments, winter burn treatments had highest height of the stand but the height was only statistically greater than the summer burn, as were the no burn and spring treatments.

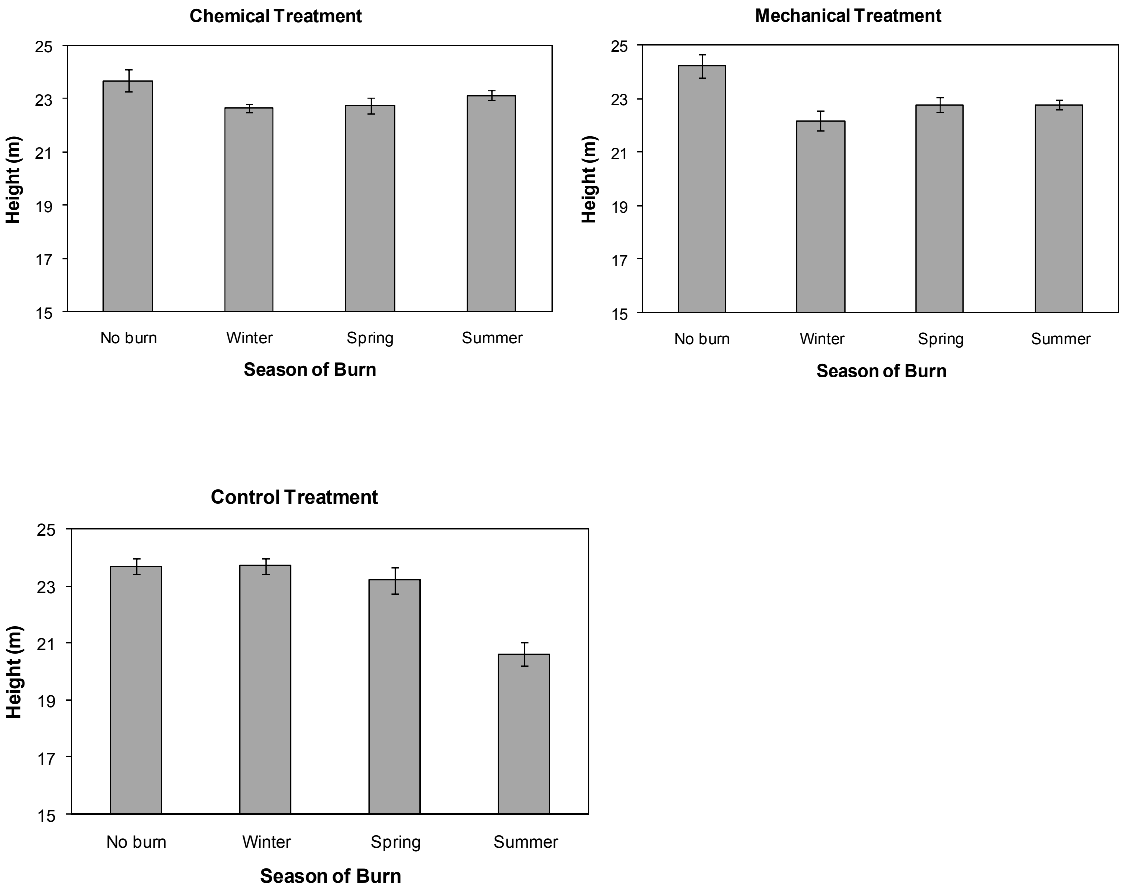


Fig 5.1.2. Average longleaf pine height (m) by burning treatment at different level of supplemental hardwood treatments. Bars represent standard errors of the means

When the individual biomass in kilograms of each longleaf pine tree encountered within each of the 0.04 ha measurement plots was analyzed using ANOVA, the supplemental treatments had no effects on longleaf pine tree biomass but the burning treatments had significant effects on the tree biomass (Table 5.1.3).

Table 5.1.3. Testing the effects of supplemental hardwood and burning treatments on biomass (kg) of individual longleaf pine trees

| **Source** | **DF** | **Type III SS** | **Mean Square** | **F-value** | **Pr > F** |
|---|---|---|---|---|---|
| Supplemental | 2 | 21968.83 | 10984.42 | 0.70 | 0.4988 |
| Burning | 3 | 993012.03 | 331004.01 | 20.98 | < .0001 |
| Supplemental*Burning | 6 | 797312.46 | 132885.41 | 8.42 | < .0001 |

Since the interaction between the supplemental treatments and burning treatments was significant, the effect of burning treatments was analyzed for each level of supplemental hardwood treatment. Average longleaf tree biomass (kg) was significantly higher in no burn treatments than all the burning treatments in chemical supplemental hardwood treatments (Appendix 5.1.3). Differences among the burning treatments were not significant. In mechanical supplemental treatment, no burn treatments had the highest average biomass but was only statistically different from winter burn treatments. Differences among season of burn were not significant. In control supplemental treatments, winter burn treatments had the highest biomass and was only statistically greater than the summer burn, as were the no burn and spring burn treatments.

ANOVA showed that both the supplemental hardwood treatments and the burning treatments had no statistically significant effects on the basal area (Table 5.1.4) and the biomass (Table 5.1.5) of longleaf pine trees at stand level.

Table 5.1.4. Testing the effects of supplemental hardwood and burning treatments on stand basal area ($m^2$ $ha^{-1}$) of longleaf pine trees

| **Source** | **DF** | **Type III SS** | **Mean Square** | **F-value** | **Pr > F** |
|---|---|---|---|---|---|
| Supplemental | 2 | 0.8994 | 0.4497 | 0.10 | 0.9044 |
| Burning | 3 | 11.604 | 3.868 | 0.87 | 0.4711 |
| Supplemental*Burning | 6 | 48.783 | 8.131 | 1.83 | 0.1365 |

Table 5.1.5. Testing the effects of supplemental hardwood and burning treatments on stand biomass (Mg $ha^{-1}$) of longleaf pine trees

| **Source** | **DF** | **Type III SS** | **Mean Square** | **F-value** | **Pr > F** |
|---|---|---|---|---|---|
| Supplemental | 2 | 155.44 | 77.72 | 0.20 | 0.8164 |
| Burning | 3 | 2503.73 | 834.58 | 2.20 | 0.1146 |
| Supplemental*Burning | 6 | 2847.64 | 474.61 | 1.25 | 0.3172 |

Both the average basal area and biomass of longleaf pine stand were not significantly different among various treatments (Table 5.1.6). However, Boyer (1993) reported the no burn plots as having the highest basal areas in the same plots. The basal area in all the burning treatments (Table 5.1.6) were higher than the average basal area, 22.2 $m^2$ $ha^{-1}$ in the same experimental plots reported by Kush et al. (2000) for 1996 measurements.

Table 5.1.6. Effects of burning treatments on average basal area and biomass of longleaf pine at stand level

| Season of burn | Basal area ($m^2$ $ha^{-1}$) | Biomass (Mg $ha^{-1}$) |
|---|---|---|
| No burn | 27.60 | 204.60 |
| Winter | 26.95 | 190.71 |
| Spring | 26.48 | 187.50 |
| Summer | 26.07 | 180.91 |

The basal area of longleaf pine stand was the highest in the no burn treatments (Fig. 5.1.3). Among the prescribed fire treatments the winter burn treatments resulted in the highest basal area and the summer burn resulted in the lowest amount of basal area. The spring prescribed burn resulted in the basal area that was between winter and summer fires. This result is almost similar to that obtained by Boyer (1993) where he reported spring burning treatments had highest basal area per acre, but the difference between winter and spring burn treatments were statistically insignificant (similar to the one obtained in this study).

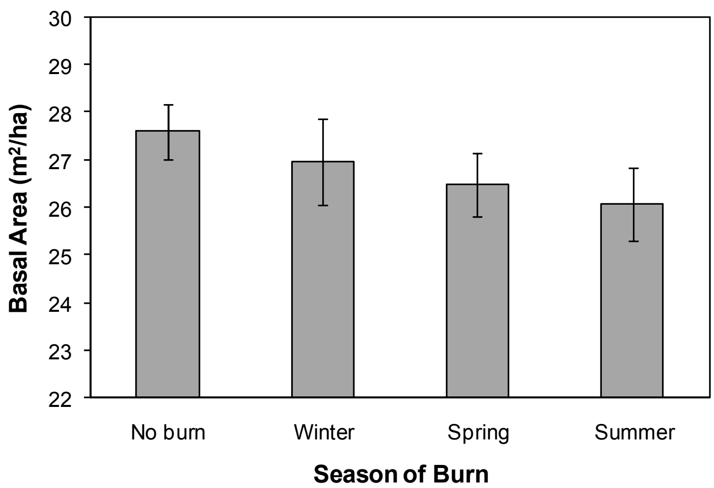


Fig 5.1.3. Basal area ($m^2$ $ha^{-1}$) of longleaf pine stand in different prescribed fire season. Bars represent standard errors of the means

The biomass (Mg ha$^{-1}$) of the longleaf pine stand was the highest in the no burn treatments (Fig. 5.1.4). Among the prescribed fire treatments the winter burn plots had the highest biomass and the summer burn resulted in the lowest amount of biomass. The result for biomass was similar to that for the stand basal areas.

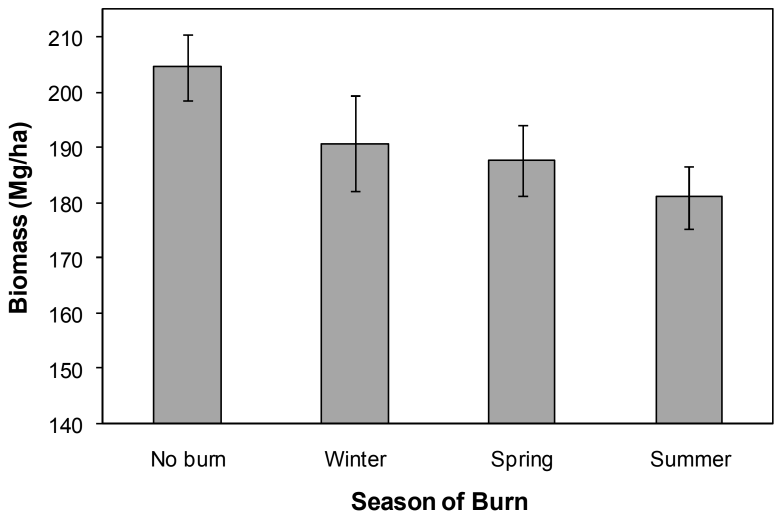


Fig 5.1.4. Biomass (Mg ha$^{-1}$) of longleaf pine stands in different prescribed fire season. Bars represent standard errors of the means

In many studies carbon content in biomass is estimated by assuming the carbon content of dry biomass to be a constant 50% by weight (Brown, 1986; IPCC, 1996; Schultz, 1997; Montagnini and Porras, 1998; Kraenzel et al., 2003; Zabek and Prescott, 2006). Since aboveground carbon content of longleaf pine trees (Mg C ha$^{-1}$) in this study was also estimated by multiplying the biomass of longleaf trees by standard coefficient of 0.5, the results were identical to that obtained with longleaf pine tree biomass (Fig 5.1.5).

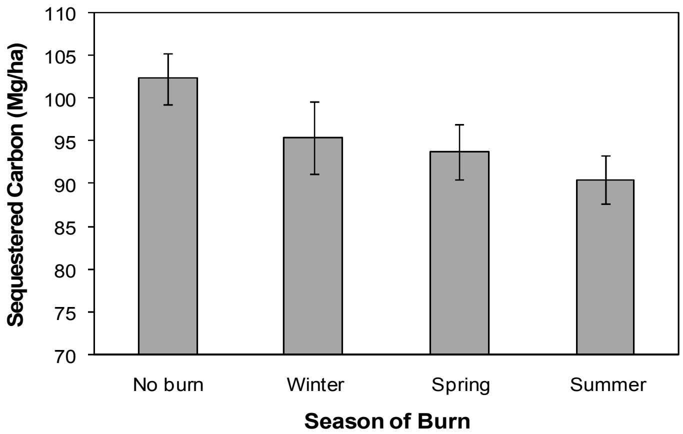


Fig 5.1.5. Sequestered carbon (Mg $ha^{-1}$) in longleaf pine stands in different prescribed fire season. Bars represent standard errors of the means

## 5.2. Understory Vegetation Dry-weight

The dry-weight of the destructively collected samples of all the five understory components was determined. Square root transformation on response variables (dry-weights) was applied to fix non-normality in the data and stabilize the non-constant variance. Two-factors ANOVA (using understory components as blocking variable) results showed that the supplemental hardwood treatments had no statistically significant effects on the dry-weight of the understory components of the longleaf pine stand, while burning treatments had significant effects (Table 5.2.1).

Table 5.2.1. Testing the effects of supplemental hardwood and burning treatments on dry-weight (gm) of understory components in longleaf pine stand

| Source | DF | Type III SS | Mean Square | F-value | Pr > F |
|---|---|---|---|---|---|
| Supplemental | 2 | 180.51417 | 90.25708 | 1.66 | 0.1940 |
| Burning | 3 | 1930.96839 | 643.65613 | 11.82 | < .0001 |
| Supplemental*Burning | 6 | 298.37395 | 49.22899 | 0.90 | 0.4938 |
| Component | 4 | 33469.01492 | 8367.25373 | 153.61 | < .0001 |

The interaction between the supplemental and burning treatments was not significant and low $p$-value of understory components indicates that blocking variable seemed to be effective in ANOVA (Table 5.2.1). No burn plots had the highest amount of dry-weight of the total understory components in the longleaf pine stand. The Tukey-Kramer test showed that the dry-weight of the no burn treatment plots was significantly different from all the three burning treatments (summer, spring, and winter) plots whereas, the dry-weight of the three burning treatments plots did not differ significantly among each other (Fig. 5.2.1f). Similarly, Dunnett's $t$ test showed that the dry-weight of the control (no burn) treatment plots was significantly different from all the three burning treatments.

Dry-weight of all the different understory components was plotted with respect to different burning treatments. Winter burn plots had the highest amount of dry-weight of grasses followed by spring burn plots and no burn had the least amount of dry-weight (Fig. 5.2.1a). Spring burn plots had the highest amount of dry-weight for the forbs component and no burn plots had the least amount (Fig. 5.2.1b). No burn plots had the highest amount of dry-weight of litter and winter burn plots had the least amount of litter (Fig. 5.2.1c). Kush et al. (2000) reported the similar trends in understory biomass by burning treatment. Their results showed that no burn plots had significantly higher total biomass than burned plots, while the total biomass values were close to each other for all burning treatments. Here, in the woody vines component, no burn had the highest amount of dry-weight of the vines and summer burn plots had the least amount of dry-weight (Fig. 5.2.1d). Similarly, no burn plots had the highest and summer burn plots had the least amount of dry-weight of shrubs (Fig. 5.2.1e). The amounts of dry-weight in the understory components are given in Appendix 5.2.1.

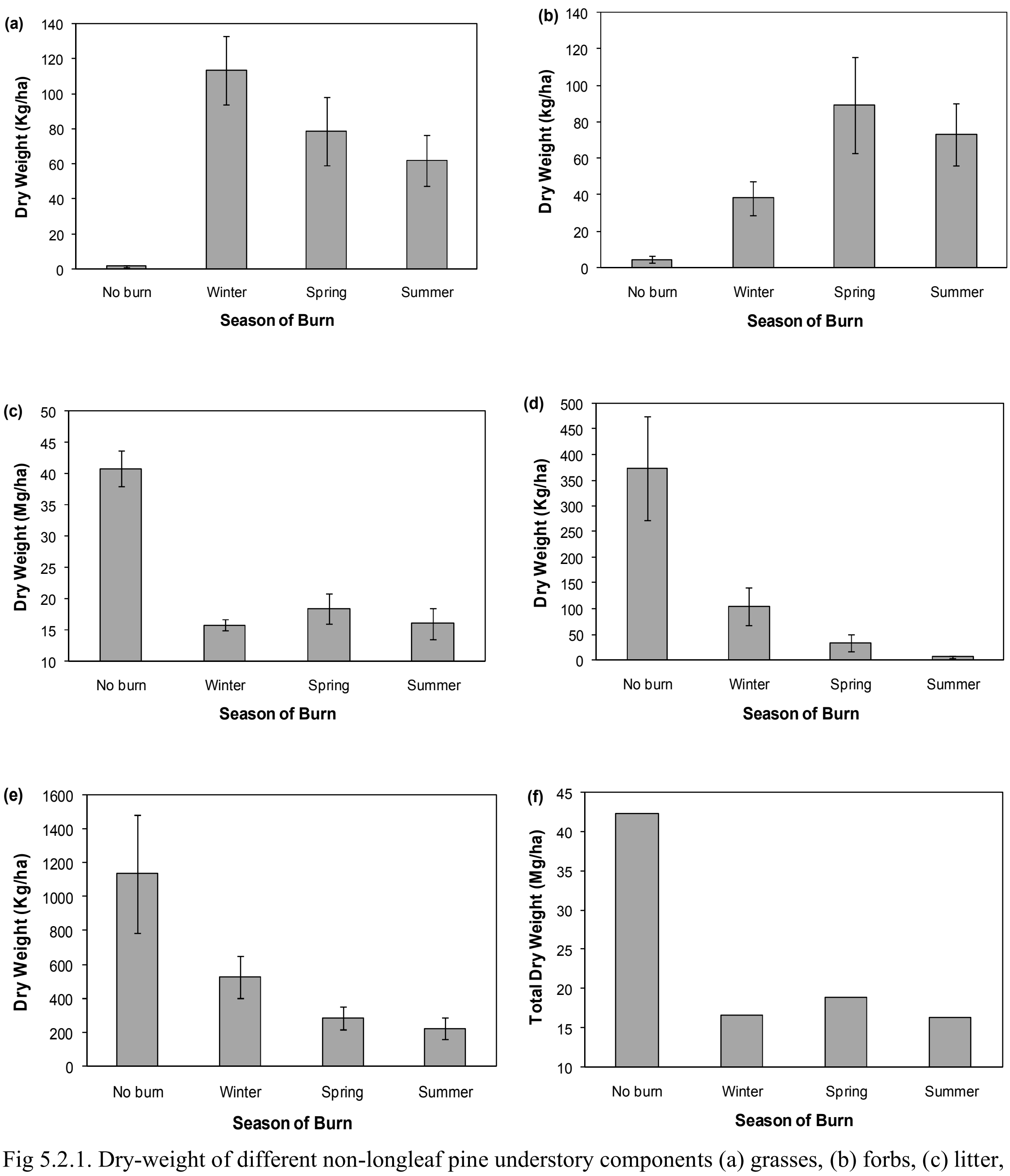


Fig 5.2.1. Dry-weight of different non-longleaf pine understory components (a) grasses, (b) forbs, (c) litter, (d) woody vines, and (e) shrubs and (f) total dry-weight of all understory components. Bars represent standard errors of the means

## 5.3. Understory Vegetation Carbon

The litter component had the highest percentage of carbon in the collected samples and the grass component had the least percentage of carbon (Table 5.3.1 and Fig. 5.3.1).

Table 5.3.1. Maximum and minimum carbon percentage in the non-longleaf pine understory vegetation and litter samples

| Understory components | Carbon percentage (%) | | Range |
|---|---|---|---|
| | Min. | Max. | |
| Grasses | 35.37 | 49.69 | 14.3 |
| Forbs | 36.43 | 51.94 | 15.5 |
| Litter | 36.00 | 55.63 | 19.6 |
| Woody vines | 44.03 | 52.39 | 8.4 |
| Shrubs | 44.71 | 54.87 | 10.2 |

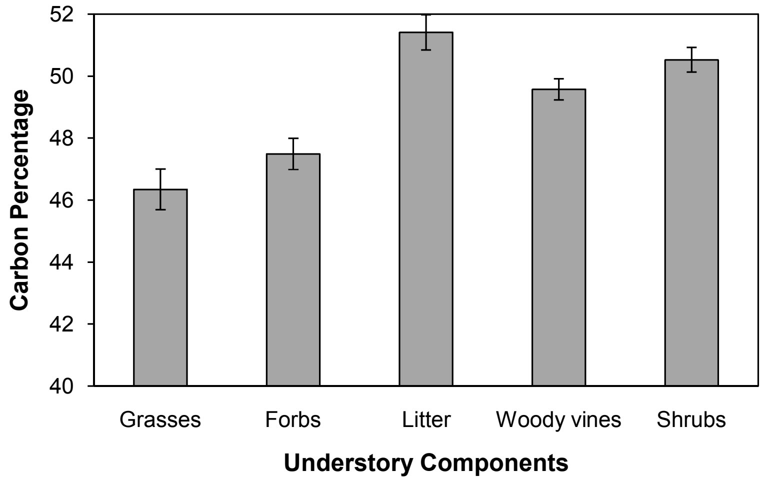


Fig 5.3.1. Mean carbon percentage in different understory components in longleaf pine stand. Bars represent standard errors of the means

The destructively collected samples of shrubs had the second largest percentage of carbon followed by woody vines and then by forbs.

As expected, the effects of the supplemental and burning treatments on carbon content of the understory components of the longleaf pine stand were similar to the ones obtained for dry-weight of the understory components. The supplemental hardwood treatments had no effects on the carbon content of the understory components but the burning treatments had significant effects on the carbon content (Table 5.3.2).

Table 5.3.2. Testing the effects of supplemental hardwood and burning treatments on carbon content of understory components in longleaf pine stand

| **Source** | **DF** | **Type III SS** | **Mean Square** | **F-value** | **Pr > F** |
|---|---|---|---|---|---|
| Supplemental | 2 | 78.70672 | 39.35336 | 1.44 | 0.2396 |
| Burning | 3 | 991.36005 | 330.45335 | 21.11 | < .0001 |
| Supplemental*Burning | 6 | 150.55406 | 25.09234 | 0.92 | 0.4826 |
| Component | 4 | 17374.83088 | 4343.70772 | 159.16 | < .0001 |

The interaction between the supplemental and burning treatments was not significant and low *p*-value of understory components indicates that blocking by understory component variable seemed to be effective in ANOVA (Table 5.3.2). The carbon content of the control treatment (no burn) plots was significantly higher than the carbon content in each of the three burning treatments (summer, spring, and winter) plots and did not differ significantly among the three burning treatments (Tukey-Kramer test). Similarly, Dunnett's *t* test showed that the carbon content of the control (no burn) treatment was significantly different from each of the three burning treatments.

Carbon stored in all the four understory vegetation types and litters was plotted by different burning treatments (Fig. 5.3.2). As expected, carbon stored in all the understory components had similar trend as the dry-weight of those same understory components.

Carbon stored in grasses was highest in the winter burn plots and least in no burn plots (Fig. 5.3.2a). Spring had burn the second highest amount of carbon stored followed by summer burn plots for grasses component. In the forbs component, spring burn plots had the highest amount of carbon stored followed by summer burn plots and no burn plots having the least carbon stored (Fig. 5.3.2b). However, the carbon stored in the litter, woody vines, and shrubs reached the highest amount in no burn plots (Fig. 5.3.2c, d, and e). Among the burn treatments, spring burn had the highest and summer burn plots had the least carbon stored in litter (Fig. 5.3.2c). In the woody vines component winter burn had the highest amount of carbon stored and summer burn plots had the least amount of carbon stored (Fig. 5.3.2d). Similarly for the shrubs, there was highest amount of carbon stored in no burn plots, and summer burn plots had least stored carbon (Fig. 5.3.2e). Total carbon stored in all the understory components was highest in no burn plots followed by winter burn plots with the summer burn plots having the least amount of carbon stored (Fig. 5.3.2f). The amounts of carbon stored in the understory components are given in Appendix 5.3.1.

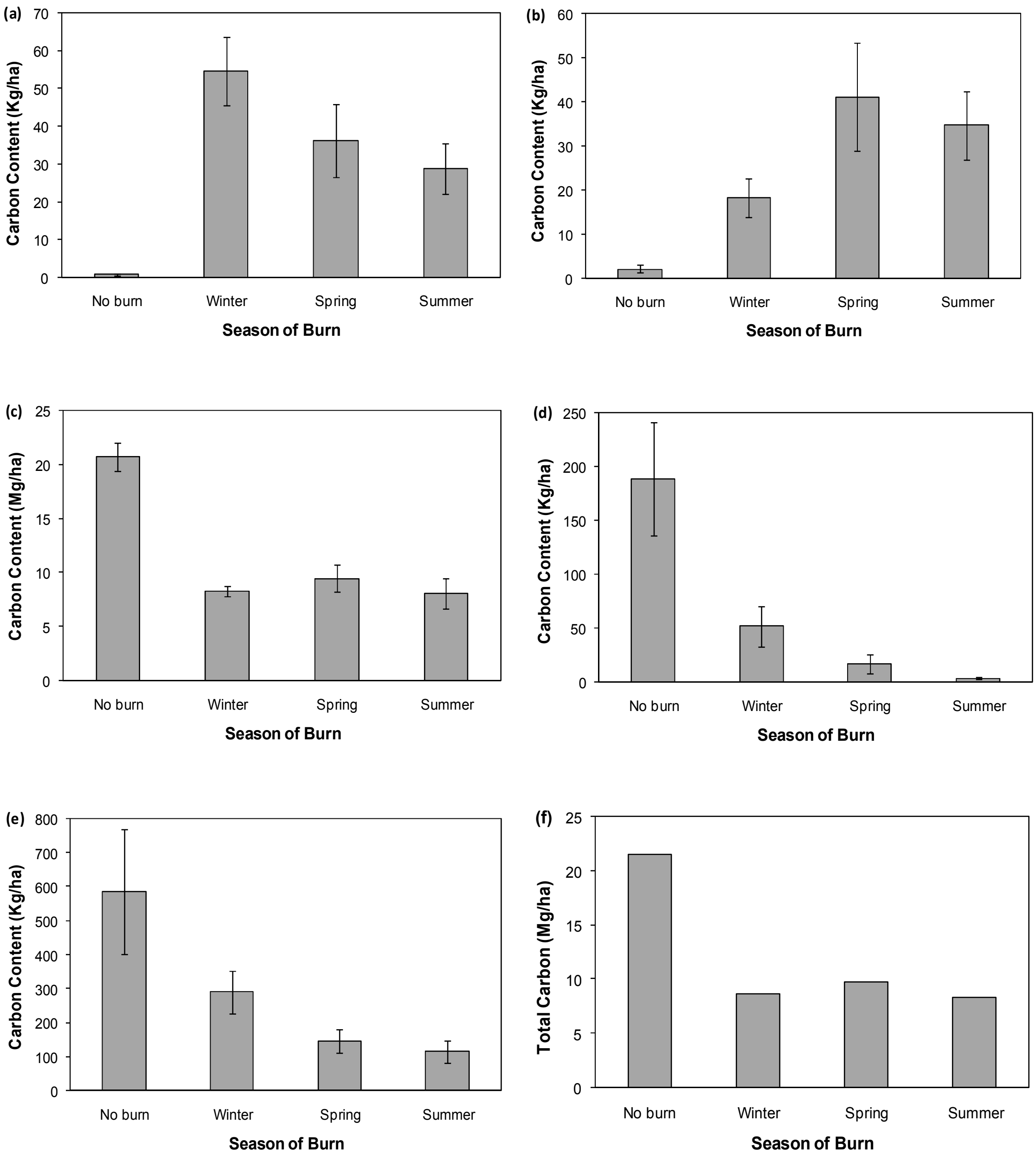


Fig 5.3.2. Carbon stored in different non-longleaf pine understory components (a) grasses, (b) forbs, (c) litter, (d) woody vines, and (e) shrubs and (f) total carbon content of all understory components. Bars represent standard errors of the means

## 5.4. Soil Carbon

### 5.4.1. First Year Sampling (2006)

Carbon content in the soil in the longleaf pine forest was expressed as percentage carbon. ARCSIN transformation of the square-root of carbon percentage was done to fix non-normality and stabilize the non-constant variance since the soil carbon values were in percentages (Anscombe, 1947; Kuehl, 2000). ANOVA results showed that the supplemental hardwood treatments had no significant effects on soil carbon, whereas the burning treatments had significant effects on the soil carbon (Table 5.4.1.1).

Table 5.4.1.1. Testing the effects of supplemental hardwood and burning treatments on soil carbon in longleaf pine stand in 2006

| **Source** | **DF** | **Type III SS** | **Mean Square** | **F-value** | **Pr > F** |
|---|---|---|---|---|---|
| Supplemental | 2 | 0.000335 | 0.000167 | 0.44 | 0.6433 |
| Burning | 3 | 0.003070 | 0.001023 | 2.71 | 0.0495 |
| Supplemental*Burning | 6 | 0.001420 | 0.000237 | 0.63 | 0.7082 |
| Depth | 2 | 0.039298 | 0.019649 | 52.05 | <.0001 |

The interaction between the supplemental and burning treatments was not significant and low *p*-value of depths of soil layer indicated that blocking variable seemed to be effective in ANOVA (Table 5.4.1.1). Carbon content in all layers combined was significantly different between the no burn and spring burn plots while other treatments did not differ from each other (Table 5.4.1.2). Similarly, when the control treatment (no burn) was compared with other three burning treatments (summer, spring, and winter) using Dunnett's t test the results were the same i.e. carbon content between no burn-spring burn pair was significantly different.

The effects of fire on carbon (C) were limited primarily to the upper 0.1 m of the mineral soil, with little change apparent in the depth below 0.1 m (Table 5.4.1.2). No burn plots had the highest carbon in all the three soil depth layers (0-0.1 m, 0.1-0.2 m, and below 0.2 m) of mineral soil. Binkley et al. (1992) reported similar results for carbon in the forest floor from a study of 30-year cumulative effects of prescribed fires at intervals of 1, 2, 3, and 4 year in a loblolly and longleaf pine forest in the Coastal Plain of South Carolina. They found the forest floor contained much more carbon and nitrogen per unit area in control plots than in 1-year and 2-year burn interval plots. However, Tilman et al. (2000) reported that belowground carbon at 0-0.2 m was lower in suppressed stands (stands experiencing 0, 1, or 2 fires in 35 years) than those in high fire frequency stands (stands experiencing from 16 to 28 fires in 35 years). Among the burning treatments, summer burn plots had the highest carbon content in upper 0.1 m layer (Table 5.4.1.2).

Table 5.4.1.2. Mean carbon content in the soil (g $kg^{-1}$) at three different depths in different burning season in 2006 and standard errors of the means

| **Burning treatment** | **Soil layer** | **Carbon (g $kg^{-1}$)** |
|---|---|---|
| No Burn | 0-0.1 m | 21.701 ± 2.82 [a] |
| | 0.1-0.2 m | 13.443 ± 1.91 |
| | Below 0.2 m | 8.724 ± 1.11 |
| Winter | 0-0.1 m | 16.964 ± 1.83 [a b] |
| | 0.1-0.2 m | 11.197 ± 1.14 |
| | Below 0.2 m | 7.551 ± 0.76 |
| Spring | 0-0.1 m | 15.179 ± 1.26 [b] |
| | 0.1-0.2 m | 9.242 ± 0.72 |
| | Below 0.2 m | 8.308 ± 1.35 |
| Summer | 0-0.1 m | 20.111 ± 2.84 [a b] |
| | 0.1-0.2 m | 11.225 ± 1.35 |
| | Below 0.2 m | 7.206 ± 0.45 |

Means within horizons across burning treatments with the same letter do not differ significantly at 0.05 level of significance (Tukey-Kramer test)

### 5.4.2. Second Year Sampling (2007)

Carbon content in the soil in the longleaf pine stand was expressed as percentage carbon. It was necessary to apply the ARCSIN square-root transformation to the 2007 carbon percentage data to correct non-normality and stabilize the non-constant variance in the data. Both the supplemental hardwood treatments and burning treatments had significant effects on soil carbon (Table 5.4.2.1)

Table 5.4.2.1. Testing the effects of supplemental hardwood and burning treatments on soil carbon in longleaf pine stand in 2007

| **Source** | **DF** | **Type III SS** | **Mean Square** | **F-value** | **Pr > F** |
|---|---|---|---|---|---|
| Supplemental | 2 | 0.002660 | 0.001330 | 6.02 | 0.0035 |
| Burning | 3 | 0.001858 | 0.000620 | 2.80 | 0.0441 |
| Supplemental*Burning | 6 | 0.003226 | 0.000538 | 2.43 | 0.0313 |
| Depth | 2 | 0.066220 | 0.033110 | 149.83 | <.0001 |

The interaction between the supplemental and burning treatments was significant and hence the effects of supplemental hardwood treatments on the soil carbon were examined for each level of burning treatment and vice versa. Low *p*-value of depths of soil layer indicated that blocking variable seemed to be effective in ANOVA (Table 5.4.2.1).

When the effects of burning treatments were analyzed within each supplemental hardwood treatment, no burn plots had the highest amount of carbon stored in the soil for chemical and control plots of supplemental hardwood treatments (Table 5.4.2.2). Soil carbon was the highest in control × no burn plots. Within chemical supplemental hardwood treatment plots, winter burn plots had the highest soil carbon among the burn treatments with summer having the least carbon. However, within mechanical supplemental hardwood treatment plots, summer burn plots had the highest carbon. Tukey-Kramer test for multiple comparisons did not pick up any significant differences in soil carbon content between any pair of burning treatments within each level of supplemental hardwood treatment. However, when no burn treatment (control) was compared with three different burning treatments within each level of supplemental hardwood treatment using Dunnett's *t* test, soil carbon content in no burn plots was significantly different from that in summer burn plots within chemical supplemental hardwood treatment (Table 5.4.2.2). Similarly, within control supplemental hardwood treatments, soil carbon content in no burn plots was significantly different from that in spring burn plots (Dunnett's *t* test).

Table 5.4.2.2. Mean carbon content in the soil (g $kg^{-1}$) at different burning treatments within each level of supplemental hardwood treatment in 2007

| **Supplemental hardwood treatments** | **Burning treatments** | **Soil layer** | **Carbon (g $kg^{-1}$)** |
|---|---|---|---|
| Chemical | No burn | 0-0.1 m | 23.904 [a] |
| | | 0.1-0.2 m | 11.896 |
| | | Below 0.2 m | 7.618 |
| | Winter | 0-0.1 m | 18.880 [a b] |
| | | 0.1-0.2 m | 9.445 |
| | | Below 0.2 m | 6.563 |
| | Spring | 0-0.1 m | 17.880 [a b] |
| | | 0.1-0.2 m | 9.187 |
| | | Below 0.2 m | 6.648 |
| | Summer | 0-0.1 m | 16.084 [b] |
| | | 0.1-0.2 m | 9.104 |
| | | Below 0.2 m | 6.049 |
| Mechanical | No burn | 0-0.1 m | 15.820 [a] |
| | | 0.1-0.2 m | 7.462 |
| | | Below 0.2 m | 5.313 |
| | Winter | 0-0.1 m | 17.857 [a] |
| | | 0.1-0.2 m | 8.701 |
| | | Below 0.2 m | 5.702 |
| | Spring | 0-0.1 m | 18.041 [a] |
| | | 0.1-0.2 m | 8.134 |
| | | Below 0.2 m | 5.005 |
| | Summer | 0-0.1 m | 19.977 [a] |
| | | 0.1-0.2 m | 9.131 |
| | | Below 0.2 m | 6.757 |
| Control | No burn | 0-0.1 m | 26.173 [a] |
| | | 0.1-0.2 m | 14.778 |
| | | Below 0.2 m | 8.144 |
| | Winter | 0-0.1 m | 17.599 [a b] |
| | | 0.1-0.2 m | 10.247 |
| | | Below 0.2 m | 7.021 |
| | Spring | 0-0.1 m | 17.665 [b] |
| | | 0.1-0.2 m | 8.005 |
| | | Below 0.2 m | 6.7885 |
| | Summer | 0-0.1 m | 25.497 [a b] |
| | | 0.1-0.2 m | 11.274 |
| | | Below 0.2 m | 7.185 |

Means of burning treatments within horizons across each supplemental hardwood treatment with the same letter do not differ significantly at 0.05 level of significance (Dunnett's *t* test).

The effects of supplemental hardwood treatments on soil carbon content were examined within each level of burning treatment using Tukey-Kramer test for multiple comparisons to test any significant differences in soil carbon content between three pairs of supplemental hardwood treatments within each level of burning treatment. Within no burn treatment plots, soil carbon content in the control supplemental hardwood treatment plots was significantly different from mechanical supplemental hardwood treatment plots. No other pairs of treatments were significant within summer, spring or winter burning treatments (Table 5.4.2.3). Results for Dunnett's *t* tests yielded the same results between the control and burning treatments.

Table 5.4.2.3. Mean carbon content in the soil (g $kg^{-1}$) at different supplemental hardwood treatments within each level of burning treatment in 2007

| Burning treatments | Supplemental hardwood treatments | Soil layer | Carbon (g $kg^{-1}$) |
|---|---|---|---|
| No burn | Chemical | 0-0.1 m | 23.904 [a b] |
| | | 0.1-0.2 m | 11.896 |
| | | Below 0.2 m | 7.618 |
| | Mechanical | 0-0.1 m | 15.820 [a] |
| | | 0.1-0.2 m | 7.462 |
| | | Below 0.2 m | 5.313 |
| | Control | 0-0.1 m | 26.173 [b] |
| | | 0.1-0.2 m | 14.778 |
| | | Below 0.2 m | 8.1449 |
| Winter | Chemical | 0-0.1 m | 18.881 [a] |
| | | 0.1-0.2 m | 9.445 |
| | | Below 0.2 m | 6.563 |
| | Mechanical | 0-0.1 m | 17.857 [a] |
| | | 0.1-0.2 m | 8.701 |
| | | Below 0.2 m | 5.702 |
| | Control | 0-0.1 m | 17.599 [a] |
| | | 0.1-0.2 m | 10.247 |
| | | Below 0.2 m | 7.021 |
| Spring | Chemical | 0-0.1 m | 17.880 [a] |
| | | 0.1-0.2 m | 9.187 |
| | | Below 0.2 m | 6.648 |
| | Mechanical | 0-0.1 m | 18.041 [a] |
| | | 0.1-0.2 m | 8.134 |
| | | Below 0.2 m | 5.005 |
| | Control | 0-0.1 m | 17.665 [a] |
| | | 0.1-0.2 m | 8.005 |
| | | Below 0.2 m | 6.788 |
| Summer | Chemical | 0-0.1 m | 16.084 [a] |
| | | 0.1-0.2 m | 9.104 |
| | | Below 0.2 m | 6.049 |
| | Mechanical | 0-0.1 m | 19.977 [a] |
| | | 0.1-0.2 m | 9.131 |
| | | Below 0.2 m | 6.757 |
| | Control | 0-0.1 m | 25.497 [a] |
| | | 0.1-0.2 m | 11.274 |
| | | Below 0.2 m | 7.185 |

Means of supplemental hardwood treatments within horizons across each burning treatment with the same letter do not differ significantly at 0.05 level of significance (Tukey's test and Dunnett's *t* test).

### 5.4.3. Change in Soil Carbon from year 2006 to 2007

Soil carbon increased in the upper 0.1 m layer of mineral soil during one year interval time period however there was an apparent decrease in carbon in depth below 0.1 m (Fig 5.4.3.1). Although no burn plots had highest amount of carbon stored in the soil, the increase was lowest in these plots, with spring burn plots having the highest increase in soil carbon in upper 0.1 m (Fig 5.4.3.1d). Prescribed fire causes either no change or an increase in mineral soil C due to the invasion of N-fixing species after burning and causes an increase in soil C over the long-term (Johnson, 1992). Possible reasons for the increase in soil C following N fixation include (i) increased productivity and, therefore, increased organic matter input to soils, and (ii) stabilization of soil organic matter.

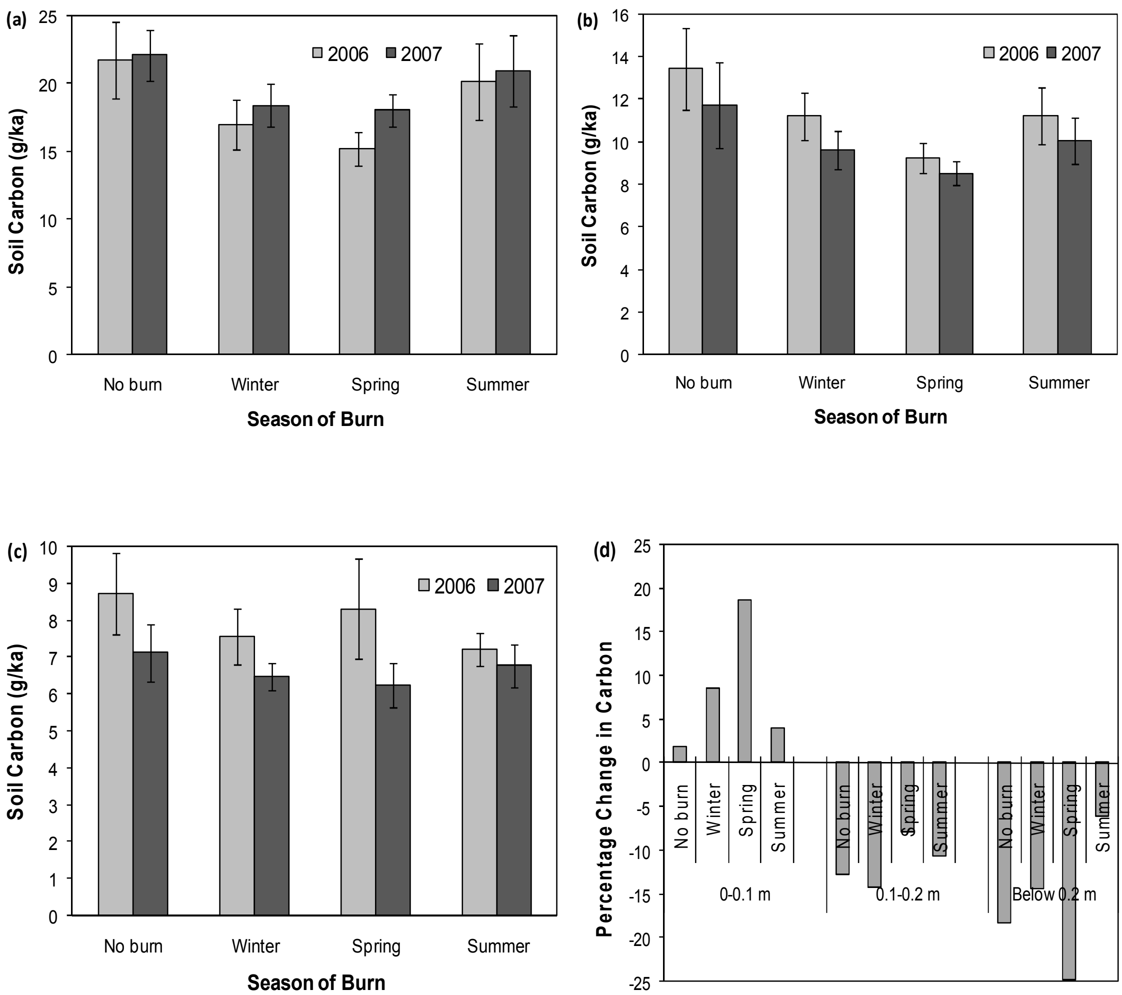


Fig 5.4.3.1. Mean carbon stored in the soil during the year 2006 and 2007 in different soil depths (a) 0-0.1 m, (b) 0.1-0.2 m, (c) below 0.2 m, (d) percentage change in carbon. Bars represent standard errors of the means

To satisfy normality and constant variance assumptions logarithmic transformation of the data was applied. ANOVA was performed on the absolute values of soil carbon differences during the one year period to examine the effects of fire on the carbon change. Both the supplemental hardwood treatments and burning treatments had no effects on the soil carbon differences (Table 5.4.3.1).

Table 5.4.3.1. Testing the effects of supplemental hardwood and burning treatments on soil carbon change during one year time period in longleaf pine stand

| Source | DF | Type III SS | Mean Square | F-value | Pr > F |
|---|---|---|---|---|---|
| Supplemental | 2 | 5.97962872 | 2.98981436 | 2.72 | 0.0713 |
| Burning | 3 | 5.42861118 | 1.80953706 | 1.64 | 0.1844 |
| Supplemental*Burning | 6 | 6.97406976 | 1.16234496 | 1.06 | 0.3944 |
| Depth | 2 | 13.07572092 | 6.53786046 | 5.94 | 0.0037 |

### 5.4.4. Repeated Measurement Design

Four-way repeated measurement designs with repeated measures on one factor (time) was performed to examine the effects of hardwood treatments and burning treatments on soil carbon content at three different soil depths during one year time period from 2006 to 2007 (Eq. 4.7.3). Repeated measure ANOVA results showed that the main effects of supplemental hardwood treatments and time periods were not significant whereas as the main effects of burning treatments and soil depths were significant (Table 5.4.4.1).

Table 5.4.4.1. Repeated measurement design: testing the effects of supplemental hardwood and burning treatments on soil carbon during one year time period at three different soil depths

| **Source** | **DF** | **Type III SS** | **Mean Square** | **F-value** | **Pr > F** |
|---|---|---|---|---|---|
| Supplemental | 2 | 0.00247474 | 0.00123737 | 1.56 | 0.2569 |
| Burning | 3 | 0.00420554 | 0.00140185 | 9.89 | 0.0008 |
| Time | 1 | 0.00067071 | 0.00067071 | 2.32 | 0.1879 |
| Depth | 2 | 0.07937551 | 0.03968776 | 16.89 | 0.0060 |
| Supplemental*Burning | 6 | 0.00356245 | 0.00059374 | 1.05 | 0.4150 |
| Supplemental*Time | 2 | 0.00073525 | 0.00036762 | 2.15 | 0.1676 |
| Burning*Time | 3 | 0.00028545 | 0.00009515 | 0.73 | 0.5512 |
| Supplemental*Depth | 4 | 0.00060046 | 0.00015011 | 0.19 | 0.9385 |
| Burning*Depth | 6 | 0.00168436 | 0.00028073 | 1.98 | 0.1326 |
| Time*Depth | 2 | 0.00064468 | 0.00032234 | 1.12 | 0.3972 |
| Supplemental*Burning*Time | 6 | 0.00089216 | 0.00014869 | 1.40 | 0.2480 |
| Supplemental*Burning*Depth | 12 | 0.00192005 | 0.00016000 | 0.28 | 0.9879 |
| Supplemental*Time*Depth | 4 | 0.00020272 | 0.00005068 | 0.30 | 0.8741 |
| Burning*Time*Depth | 6 | 0.00063439 | 0.00010573 | 0.81 | 0.5790 |
| Supplemental*Burning*Time*Depth | 12 | 0.00084456 | 0.00007038 | 0.66 | 0.7729 |

No two-way and three-way interactions were significant (Table 5.4.4.1). The main effects of burning treatments on soil carbon were examined using LSMEANS in SAS (Table 5.4.4.2).

Table 5.4.4.2. Mean carbon content in the soil (g kg$^{-1}$) during one year time period

| **Burning treatments** | **Soil Carbon (g kg$^{-1}$)** |
|---|---|
| No burn | 13.120 $^{a}$ |
| Winter | 11.158 $^{b}$ |
| Spring | 10.355 $^{b}$ |
| Summer | 11.822 $^{a\,b}$ |

Significant differences at 0.05 level of significance are denoted by different letters for each column means (Tukey test).

Three-way repeated measurement designs with repeated measures on soil depths were also performed to examine the effects of hardwood treatments and burning treatments on soil carbon contents at three different soil depths for each of the years i.e. 2006 and 2007 (Tables 5.4.4.3 and 5.4.4.4 respectively). In both years main effects of burning treatments and soil depths were significant; however no other main effect or interaction was significant.

Table 5.4.4.3. Repeated measurement design: testing the effects of supplemental hardwood and burning treatments on soil carbon in 2006 at three different soil depths

| **Source** | **DF** | **Type III SS** | **Mean Square** | **F-value** | **Pr > F** |
|---|---|---|---|---|---|
| Supplemental | 2 | 0.00041167 | 0.00020584 | 0.37 | 0.7015 |
| Burning | 3 | 0.00327110 | 0.00109037 | 8.72 | 0.0014 |
| Depth | 2 | 0.03313522 | 0.01656761 | 8.11 | 0.0270 |
| Supplemental*Burning | 6 | 0.00157381 | 0.00026230 | 0.63 | 0.7030 |
| Supplemental*Depth | 4 | 0.00067839 | 0.00016960 | 0.30 | 0.8696 |
| Burning*Depth | 6 | 0.00188800 | 0.00031467 | 2.56 | 0.0689 |
| Supplemental*Burning*Depth | 12 | 0.00146564 | 0.00012214 | 0.29 | 0.9856 |

Table 5.4.4.4. Repeated measurement design: testing the effects of supplemental hardwood and burning treatments on soil carbon in 2007 at three different soil depths

| **Source** | **DF** | **Type III SS** | **Mean Square** | **F-value** | **Pr > F** |
|---|---|---|---|---|---|
| Supplemental | 2 | 0.00266041 | 0.00133020 | 3.33 | 0.0707 |
| Burning | 3 | 0.00185794 | 0.00061931 | 4.08 | 0.0224 |
| Depth | 2 | 0.06622008 | 0.03311004 | 58.75 | 0.0001 |
| Supplemental*Burning | 6 | 0.00322635 | 0.00053772 | 2.37 | 0.0500 |
| Supplemental*Depth | 4 | 0.00011287 | 0.00002822 | 0.07 | 0.9897 |
| Burning*Depth | 6 | 0.00036812 | 0.00006135 | 0.40 | 0.8664 |
| Supplemental*Burning*Depth | 12 | 0.00121460 | 0.00010122 | 0.45 | 0.9325 |

# 6. CONCLUSIONS

The first objective of this study was to develop a better understanding between the above ground biomass and carbon sequestration in a natural longleaf pine ecosystem. Since the longleaf pine ecosystem is a fire-maintained ecosystem, this study evaluated the response of a natural longleaf pine ecosystem to biennial seasonal burning and supplemental hardwood control. Four level of burning treatments (winter, spring, summer, and no burn) were applied in combination with three levels of supplemental hardwood treatments (chemical, mechanical, and control).

Several significant changes had occurred in response to the different seasonal burns in the study area since reported previously by Boyer (1993) and Kush et al. (2000). The burning treatments significantly affected the DBH and diameter of the longleaf pine stand. Although no apparent significant effect of supplemental hardwood control was noticed on DBH, height, basal area, and biomass of the overstory longleaf pine trees, it had certainly influenced hardwood development in young, naturally established longleaf pine stand. Average DBH and height of longleaf pine trees were greater on no-burn plots. Among the burning treatments, spring burn plots had the highest DBH of longleaf trees and summer burn plots had the highest height of longleaf pine trees. DBH and height of longleaf trees had increased significantly than the last report by Boyer (1993) in the same study area. Burning treatments were found to have significant effects on the individual biomass of longleaf pine trees at individual tree level although no significant effects were observed on biomass of longleaf pine trees at stand level. No burn plots had the highest biomass of longleaf trees and among the burning treatments, the spring burn plots resulted in the highest biomass in chemically and mechanically treated plots. No significant effects of burning and supplemental hardwood treatments on the basal area and biomass of the longleaf trees at stand level were observed. There was significant increase in basal area and biomass of longleaf pine trees than reported by Kush et al. (2000).

Above-ground carbon sequestered in longleaf pine trees was greatest in the no burn plots but there was little difference in carbon sequestered between the burning treatments. Burning treatments significantly affected the understory biomass. Significantly higher total understory biomass was recorded in the no burn plots, but the total understory biomass in the burning treatments did not differ significantly among each other. The supplemental hardwood treatments had no effect on the carbon content of the understory components while the burning treatments did significantly affect the carbon content in understory biomass. Significantly higher total carbon was documented in the no burn plots. Among the burning treatments, winter burn plots had the highest total carbon, which is related to the quantity of woody vines and shrubs on that treatment. Grass biomass carbon was highest in the winter burn plots and least in the no burn plots. Spring burn plots had the highest amount of carbon in forb and no burn plots having the least carbon stored. However, the biomass carbon stored in no burn plots reached the highest amount in the litter, woody vine, and shrub components of the understory vegetation. Among the burning treatments, spring burn had the highest biomass carbon stored in litter and winter burn plots had the highest in biomass carbon in woody vines and shrubs.

Although the total above-ground biomass carbon stored in understory vegetation and litter was significantly higher in the no burn plots, it is not intended to recommend exclusion of fire from the longleaf pine ecosystem. The longleaf pine ecosystem, being a fire-maintained ecosystem, requires fire for its perpetuity and fire exclusion may eventually lead to disappearance of longleaf trees altogether and result in encroachment by hardwood species and other pines. The estimated rates of carbon storage that might result in the absence of burning are not sustainable in the long term. The increased carbon storage associated with fire suppression represents an accumulation of fuel that might result in catastrophic, stand-destroying fires, especially during droughts.

This study did not examine the compositions and structures of the understory plant communities and their response to biennial season burn. The absence of fire can have a negative impact on understory species richness in longleaf pine ecosystem. On the same study site, Kush et al. (2000) reported total of 114 species of understory plants (grasses, forbs, legumes, and woody vines) in winter burn plots compared to 104 species in spring burn and summer burn plots and 84 species in no burn plots in longleaf pine forests. They also reported that winter burning produced about 92% more legume biomass than did the control burning and underscored the implication of biennial burning in wildlife management goal to produce the most biomass in legumes for wildlife food. Since winter burn plots were found to have stored more carbon in the understory components, winter burning can be utilized to increase carbon in the understory vegetation despite the fact that there has been a push towards growing-season burning in longleaf pine ecosystem. Besides, the winter season burning is more favored to have higher understory species richness and more legume biomass.

Forest soils contain more than 70% of the terrestrial world's soil carbon pool yet little is known about the effects of prescribed fire on forest soil carbon sequestration. This study tried to assess the potential for soil carbon sequestration in a natural longleaf pine ecosystem. In 2006, burning treatments significantly affected the carbon content in the mineral soil in the upper 0.3 m layer while the supplemental hardwood treatments did not significantly affect the mineral soil carbon. The effect of biennial burning was primarily limited to the upper 0.1 m of the mineral soil with little change apparent in the depth below 0.1 m. No burn plots had the highest carbon stored in all the three soil depth layers (0-0.1 m, 0.1-0.2 m, and below 0.2 m) of mineral soil and summer burn plots had the highest carbon content among the burning treatments.

In the second year soil sampling (2007) both the supplemental hardwood treatments and burning treatments had significant effects on soil carbon and also the interaction between the supplemental and burning treatments was significant. No burn plots had the highest amount of carbon stored in the soil for chemical and control plots of supplemental hardwood treatments. Winter burn plots recorded the highest soil carbon content among burning treatment in chemical supplemental treatment plots and summer burn had the highest soil carbon in control supplemental treatment plots. Both the winter season and growing season burning are more likely to increase the soil carbon in a natural longleaf pine forest.

Increase in soil carbon was observed in the upper 0.1 m layer of mineral soil during one year time period however there was decrease in carbon in depth below 0.1 m. This increase in soil C by prescribed fire may be due to incorporation of charcoal into soil and new C inputs via post-fire $N_2$ fixation. Although no burn plots had highest amount of carbon stored in the soil, the increase was lowest in these plots with spring burn plots having the highest increase in soil carbon in upper 0.1 m layer during one year time period. The greater increase of soil C in spring burn plots as compared to no burn plots may be attributed to higher number of legume species in burning plots against no burn plots as reported by Kush et al. (2000). However, one year time interval is an insufficient period in which to measure changes in soil carbon. Implementation of prescribed burning at regular interval in these longleaf pine forests in particular may also limit the potential for soil carbon sequestration. However, this is not to suggest that no carbon will be sequestered by this ecosystem in the soil, since relatively large quantities can accumulate in aboveground and belowground (root) biomass. Further, over a longer course of time period soil carbon may begin to accumulate to levels observed in the no burn treatment plots.

Due to the subdued topography and well-drained soils characteristic of the U.S. Southern Coastal Plain, prescribed fire-caused nutrient losses to erosion and leaching are generally negligible. The soil carbon loss in the mineral soil below 0.1 m depth in the study site was very negligible. Prescribed fire has various beneficial effects which may offset any losses in productivity. However, optimizing the benefits of prescribed fire will require a broad understanding of the short- and long-term effects of burning on soil carbon sequestration and ecosystem processes. A more complete understanding of carbon trends in longleaf pine forest soils requires a longer-term examination. More precise quantification of the biogeochemical effects of prescribed fires needs to be examined over longer period of time rather than at one or two times.

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## APPENDICES

**Appendix 5.1.1. The basic statistical measures of DBH and height of the longleaf pine trees measured in the sample plots**

| Basic Statistical Measures | DBH (cm) | Height (m) |
|---|---|---|
| Maximum | 38.35 | 28.04 |
| Minimum | 4.83 | 3.35 |
| Range | 33.52 | 24.69 |
| Mean | 23.24 | 22.84 |
| Standard deviation | 5.13 | 2.97 |
| Variance | 26.33 | 8.82 |
| No. of observations | 867 | 867 |

**Appendix 5.1.2. Effects of burning treatments on average DBH (cm) and height (m) of longleaf pine stand within supplemental hardwood treatments**

| Supplemental hardwood treatments | Burning treatments | DBH (cm) | Height (m) |
|---|---|---|---|
| Chemical | No burn | 25.91 [a] | 23.68 [a] |
| | Winter | 22.56 [b] | 22.64 [a] |
| | Spring | 23.42 [a b] | 22.74 [a] |
| | Summer | 23.02 [b] | 23.13 [a] |
| Mechanical | No burn | 25.01 [a] | 24.23 [a] |
| | Winter | 22.08 [b] | 22.17 [b] |
| | Spring | 23.85 [a b] | 22.75 [a b] |
| | Summer | 23.78 [a b] | 22.77 [a b] |
| Control | No burn | 24.39 [a] | 23.68 [a] |
| | Winter | 24.74 [a] | 23.70 [a] |
| | Spring | 23.08 [a] | 23.20 [a] |
| | Summer | 19.19 [b] | 20.61 [b] |

Significant differences within a supplemental hardwood treatment at 0.05 level of significance are denoted by different letters for each column means (Tukey-Kramer test)

**Appendix 5.1.3. Effects of burning treatments on biomass (kg) of individual longleaf pine tree within supplemental hardwood treatments**

| Supplemental hardwood treatments | Burning treatments | Average longleaf tree biomass (kg) |
|---|---|---|
| Chemical | No burn | 382.96 [a] |
| | Winter | 272.73 [b] |
| | Spring | 290.86 [b] |
| | Summer | 290.51 [b] |
| Mechanical | No burn | 369.59 [a] |
| | Winter | 267.60 [b] |
| | Spring | 306.89 [a b] |
| | Summer | 303.09 [a b] |
| Control | No burn | 335.54 [a] |
| | Winter | 347.55 [a] |
| | Spring | 308.84 [a] |
| | Summer | 208.49 [b] |

Significant differences within a supplemental hardwood treatment at 0.05 level of significance are denoted by different letters for each column means (Tukey-Kramer test).

**Appendix 5.2.1. Dry-weight in different non-longleaf pine understory components at different burning treatments**

| Season of burn | Dry-weight in non-longleaf pine understory components (Kg $ha^{-1}$) | | | | | |
|---|---|---|---|---|---|---|
| | Grasses | Forbs | Litter | Woody vines | Shrubs | Total carbon |
| **No burn** | 1.40 $^{a}$ | 4.28 $^{a}$ | 40801.96 $^{a}$ | 373.16 $^{a}$ | 1135.20 $^{a}$ | 41180.80 $^{a}$ |
| **Winter** | 113.33 $^{b}$ | 38.12 $^{b}$ | 15736.33 $^{b}$ | 103.64 $^{b}$ | 521.96 $^{a\ b}$ | 15991.42 $^{b}$ |
| **Spring** | 78.64 $^{b}$ | 89.41 $^{b}$ | 18333.03 $^{b}$ | 32.38 $^{b\ c}$ | 282.59 $^{b}$ | 18533.45 $^{b}$ |
| **Summer** | 62.02 $^{b}$ | 73.12 $^{b}$ | 15970.34 $^{b}$ | 5.58 $^{c}$ | 219.35 $^{b}$ | 16111.05 $^{b}$ |

Significant differences within understory components at 0.05 level of significance are denoted by different letters for each row (Tukey-Kramer test)

**Appendix 5.3.1. Carbon stored in different non-longleaf pine understory components at different burning treatments**

| Season of burn | Carbon content in non-longleaf pine understory components (Kg ha$^{-1}$) | | | | | |
|---|---|---|---|---|---|---|
| | Grasses | Forbs | Litter | Woody vines | Shrubs | Total carbon |
| **No burn** | 0.64 $^{a}$ | 2.09 $^{a}$ | 20701.31 $^{a}$ | 188.57 $^{a}$ | 584.65 $^{a}$ | 21477.26 $^{a}$ |
| **Winter** | 54.59 $^{b}$ | 18.22 $^{b}$ | 8242.64 $^{b}$ | 51.35 $^{b}$ | 289.28 $^{a\,b}$ | 8656.08 $^{b}$ |
| **Spring** | 36.25 $^{b}$ | 41.09 $^{b}$ | 9459.04 $^{b}$ | 16.16 $^{b}$ | 143.69 $^{b}$ | 9696.23 $^{b}$ |
| **Summer** | 28.71 $^{b}$ | 34.64 $^{b}$ | 8066.05 $^{b}$ | 2.74 $^{b}$ | 114.15 $^{b}$ | 8246.29 $^{b}$ |

Significant differences within understory components at 0.05 level of significance are denoted by different letters for each row (Tukey-Kramer test)