

FIRE EFFECTS AND LITTER ACCUMULATION DYNAMICS
IN A MONTANE LONGLEAF PINE ECOSYSTEM

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IN A MONTANE LONGLEAF PINE ECOSYSTEM

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Dedication

This Thesis is dedicated to my family and friends
for their support over many long years.

I would like to recognize my parents and thank them
for their devotion, dedication, love, and faith throughout my life,
and especially over the last two years.

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ABSTRACT

A common obstacle for public land managers developing fire management plans in the eastern and southern United States is the lack of quantitative information on historic fire regimes and the effects that varying fine fuel loadings can produce. Despite the ecological importance of litter, little is known about the effects of litter accumulation and decay rates in the montane longleaf pine (*Pinus palustris* Mill.) region. Historic fire information helps to set target goals for wildland fire practices and fuels management. In this thesis, four centuries of past fire regimes on the Talladega National Forest in northeastern Alabama are described along with litter dynamics of recent prescribed burning practices. Seasonally distinguishable (e.g., dormant, early, and late growing season) fire events from 372 tree-ring dated fire scars on 50 longleaf pine remnants and live trees were used to reconstruct past fire regimes. Litter accumulation, combined litter measurements, and model estimates were used to derive decay constants that characterize montane longleaf pine. The fire regime prior to the early 19th century was characterized by a mean fire interval (MFI) of 3 years. During this time period most fires were small, low severity burns, often scarring only 1-2 trees in the sampled area; however, some fires appeared to be severe, scarring multiple trees throughout the landscape. The onset of EuroAmerican settlement in the mid-19th century changed the fire regime to 2.5 years. The number of fires decreased during the 20th century due to changes in land use, anthropogenic influences and climate-fire relationships. Litter accumulation equations were used to demonstrate temporal changes in litter loading. For example, after a fire event that consumes nearly 100 percent of the litter, about 35 percent of the litter

accumulation equilibrium is reached within 1 year, 58 percent within 2 years, and the equilibrium (99 percent accumulation) after approximately 10 years. These results can be used to determine the appropriate prescribed burning intervals for a desired fire severity.

CHAPTER 1

INTRODUCTION

The mature old growth longleaf pine (*Pinus palustris* Mill.) forests and savannas of the southeastern United States were once among the most extensive and ecologically diverse ecosystems in North America. Historically, these forests spanned nearly 40 million hectares, with a contiguous range extending from eastern Texas to southern Florida and northward to southern Virginia. Prior to Spanish arrival in the 1500s, it is estimated that pure longleaf pine stands covered about 24 million hectares and mixed longleaf/pine/hardwood stands covered another 12 million hectares (Frost, 1993). Longleaf pine was highly valued in early European settlements in the Southeast due to its beauty and wide range of uses.

The onset of EuroAmerican settlement drove a changing fire regime and hastened the decline of the once extensive longleaf ecosystem. Settlements in the 18th century were built using the surrounding resources, and demand on these resources only increased as settlements expanded through the 20th century. The lumbering and turpentine industry produced extreme stresses on the forests ability to regenerate. Crop trees for turpentine use were located throughout the entire Southeast and were typically used for many years then abandoned (Frost, 1993). Deforestation and turpentine use led to a significant decrease in the success and sustainability of the longleaf pine ecosystem. Currently, longleaf pine forests are highly fragmented and estimated at less than 1.2 million hectares (Early, 2004; Frost, 2000; Landers et al., 1995; Outcalt and Sheffield, 1996).

Historically, longleaf pine forests were maintained through both large- and small-scale disturbances. Large-scale disturbances such as catastrophic hurricanes and 100-year floods still influence the coastal landscapes of the Southeast. However, the small-scale disturbances that may have historically shaped this landscape have been dramatically altered since EuroAmerican settlement. Evidence suggests that prior to European settlement, longleaf pine forests of the Southeast coastal plains were maintained by relatively frequent occurring fires through Native American and natural (lightning) ignitions (Early, 2004). Changes in the frequency of surface fires since EuroAmerican settlement have impacted the structure and processes of longleaf pine forests. Land fragmentation due to agriculture and urbanization and fire suppression policies led to a drastic reduction in fire frequency compared to the historical frequency (Frost, 1993; Henderson, 2006; Shankman and Malcolm, 1995). Small-scale, localized fire events are necessary to maintain the open understory required for regeneration of longleaf, and ensure the sustainability of the longleaf pine ecosystem (Early, 2004; Frost, 2006; Platt and Rathburn, 1993).

The longleaf ecosystem currently extends from the southeastern coastal plains to the lower mountainous region of the Appalachians in northeastern Alabama and Georgia. Together, the area makes up the range of longleaf pine forests, however within this larger distribution there remains three distinctive ecosystems. Recent efforts to restore or preserve natural longleaf ecosystems usually include some component of prescribed burning. However, burning prescriptions in montane longleaf forests are based on data derived from studies conducted in the southeastern coastal region—a much different landscape than typically found in the mountainous region of northeastern Alabama. In

addition, burning practices are often dictated by the availability of resources, ground personnel, and prescription guidelines. Understanding the importance of recurring fire events and the effects of litter accumulation in a specific area can provide managers with a better foundation for prescribed burning plans.

The success of longleaf pine forest restoration or preservation depends on a balance of frequent small-scale disturbances to prepare the seedbed for regeneration and to reduce competition (Early, 2004; Varner and Kush, 2004). The success of longleaf pine is important; not only does it have historical significance throughout the southeastern United States, but the longleaf pine ecosystem includes diverse wildlife and plant communities which depend on the long-term viability of the ecosystem (Varner et al., 2003b). Longleaf pine is found on a variety of sites ranging from dry, rocky areas to wet, low-lying flatwoods. Like many pine species, longleaf is described as shade intolerant, but it regenerates well under mature stands of longleaf following surface fires because stands are typically open and allow ample light into the understory (Hardin et al., 2001; Henderson, 2006). Longleaf pine also has the ability to obtain many structural canopy positions (i.e., overstory, understory, and regeneration); this means that once a healthy longleaf forest has been established, with proper management, it will continue to replace itself as the dominant forest species.

In the present study, I examine the effects of small-scale surface fires and their relation to the accumulation of longleaf pine leaf litter and decay in historically logged montane longleaf pine forests in northeastern Alabama. A chronosequence (time-step) based approach was used to investigate the accumulation and decay of fine leaf litter. The purpose of this study is to provide information about the role that fine fuels play in

the behavior of fire, and the overall effects of fuel consumption at varying fire intervals. In addition, this research seeks to further elucidate the influence of recurring fire on montane longleaf pine ecosystem processes.

The study objectives include:

- 1) describing the role of fire in montane longleaf pine ecosystems;
- 2) reconstructing historical fire events from fire-scar data and modeling the pre-EuroAmerican settlement mean fire return interval of the forest;

H₀: The mean fire return interval for montane longleaf pine forests will be shorter during pre-EuroAmerican settlement than EuroAmerican settlement;

H_a: The mean fire return interval for montane longleaf pine forests will be longer during pre-EuroAmerican settlement than EuroAmerican settlement;

- 3) characterizing the fine fuel loading and decay rates in relation to fire disturbance events using a chronosequence approach;

H₀: The mass of fine fuels and decay rates increase during longer periods between fire disturbance events;

H_a: The mass of fine fuels and decay rates decreases during longer periods between fire disturbance events;

- 4) determining the fire regime that is most suitable for longleaf pine regeneration, sustainability, and fuels management.

This research aims to address questions about the dynamics of litter and decay in relation to varying fire intervals for montane longleaf pine. Furthermore, the results will

be used to model the potential effects of varying fire return intervals, which can help land managers determine when a site is too infrequently burned or additional management practices (e.g., thinning or timber stand improvement) might be required to achieve successful longleaf pine regeneration.

The primary relationship discussed throughout this study will be that of fire and leaf litter fuels. The highly volatile chemical mixture of longleaf pine needles and surrounding vegetation can influence fire behavior. This mixture can create either the most opportunistic bed for seeding, or it can have devastating mortality when combined with dry winds and high stocking levels. The overall intent of this research is to provide information, with respect to fine fuels and fire disturbance, which will increase the understanding of longleaf pine ecology and forest disturbance processes in mountainous systems.

CHAPTER 2

LITERATURE REVIEW

The Montane Longleaf Pine Ecosystem of Alabama

The earliest descriptions of the longleaf pine forests were characterized as being “of a vast and unending landscape that was full of beauty and endless timber resources” (Early 2004). The terms “flatwoods”, “sand-hills”, and “savannas” were historically used to differentiate among longleaf forests as the terrain changed. Historical photographs, witness tree studies, and written documentation of longleaf pine forests in the southwestern Appalachian Mountains (Mohr, 1896; Predmore et al., 2007; Reed, 1905) suggest open-canopy and uneven-aged stands of pure longleaf with basal areas ranging from 12 to 35 m² ha⁻¹ (Reed, 1905; Schwarz, 1907); individual longleaf trees could obtain diameters up to 100 cm and heights up to 40 m (Varner and Kush, 2004).

It was not until the late 20th century that ecologists would document longleaf pine forests as one of the most biologically diverse ecosystems on earth (Early, 2004). Today, longleaf communities are classified based on their plant and animal associations and physiographic growing regions. The three distinctive transitional zones (Figure 2.1) where longleaf remains the dominant species are: coastal (located along the southeastern coastline from Louisiana to Virginia where flooding is frequent), Piedmont (situated between the mountainous regions of the Appalachians and Smoky Mountains and the coastal longleaf region), and montane (areas of northern Alabama and Georgia at the southern-most extent of the Appalachian Mountains). Longleaf pine can be found in elevations ranging from sea level (coastal longleaf regions) to around 600 m above sea level (a.s.l.) in northern Alabama (montane regions). All types of longleaf pine forests

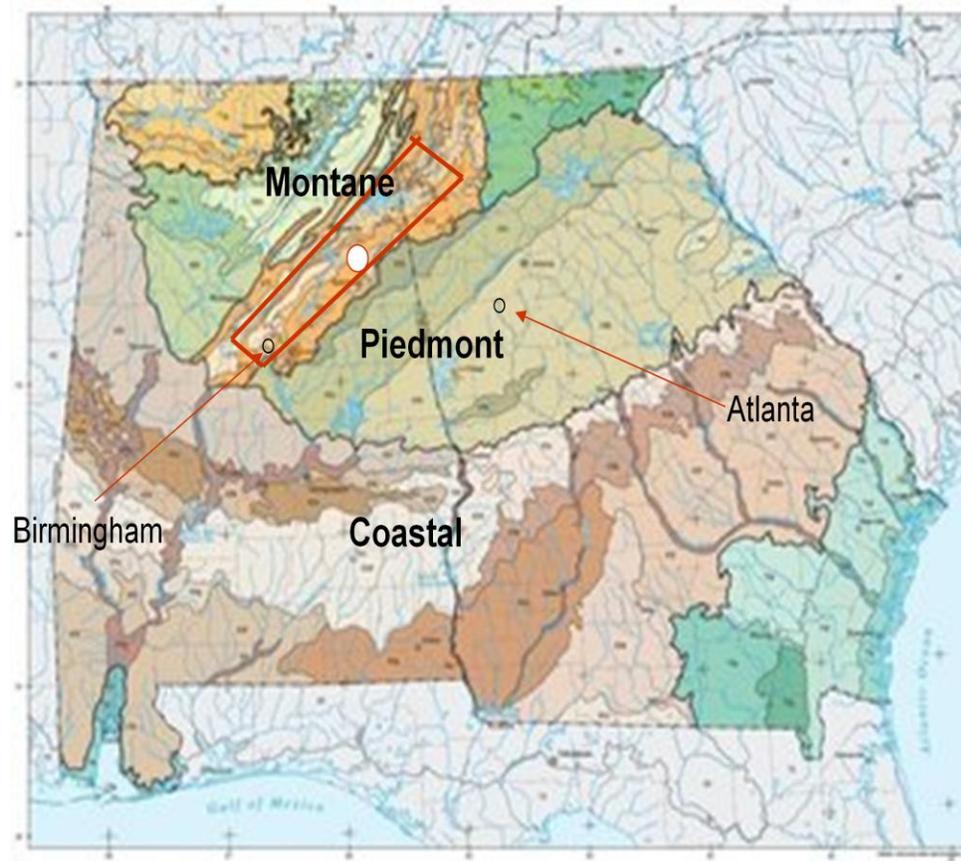


Figure 2.1. Boundaries for the three major transitional zones of longleaf pine. Site location is outlined in a white circle and located within the montane zone. This image was modified from the Ecoregions of Alabama and Geroiga (Griffith et al., 2001).

are characterized as being fire-dependent for regeneration and control of encroaching or invading species (Hardin et al., 2001).

This study will focus on the mountain longleaf region where less than 40,000 ha of montane longleaf pine ecosystem remains (Outcalt and Sheffield, 1996; Varner et al., 2003a). There is no rigorous delineation of a mountain system for montane longleaf, but it is generally confined to the Blue Ridge, Ridge and Valley, and Cumberland Plateau physiographic regions (Varner et al., 2003b). The study sites are located at the northern extent of the montane longleaf physiographic province, near Choccolocco Mountain in the Talladega National Forest, Alabama.

Physiography, topography, and geology

The Shoal Creek Ranger District of the Talladega National Forest is located within the Talladega Upland subdivision of the Piedmont ecoregion (Bailey, 1995). The Talladega Upland is heavily forested and extremely dissected. Topographic variation ranges from valleys that are periodically inundated with water throughout many months of the year to very steep slopes that are dry. Local relief of 400 m is common. Elevations range between 300 m to 700 m a.s.l., though longleaf pine is rarely found above 600 m (Varner et al., 2003b).

Geologic formations of the mountainous Piedmont ecoregion are a complex mosaic of Precambrian and Paleozoic metamorphic and igneous rocks. The Talladega Upland is characterized by Silurian to Devonian age phyllite, quartzite, slate, metasilstone, and metaconglomerate (Griffith et al., 2001). The more mountainous regions within the Talladega National Forest and its spur ridges are all formed from

sandstone and metaconglomerate, and many of these ridges are capped with Weisner quartzite.

Soils

Soils of the Talladega Upland ecoregion were formed principally by the weathering of the metamorphic and igneous rock formations. Soils contain high amounts of loose rock fragments, and outcrops of quartzite and sandstone bedrock are representative of many mountain longleaf pine ecosystems (Craul, 1965). Montane longleaf pine sites are often associated with steep uplands, on strongly dissected plateau edges, and on the low mountain areas throughout Cleburne County.

There are a greater number of soil types than forest type associations throughout the Cleburne County, indicating that different soils support similar forest types. Spivey (1974) identified 18 soil series within Cleburne County and defined them by soil types, topographic features, and land use association. The major soils corresponding to the study area of this research include Madison-Louisa and Tatum-Tallapoosa-Fruithurst associations. These two major soil associations make up 36 percent of the county and support pine and pine-hardwood forest types.

The Madison-Louisa soil association consists of Ultisols and Inceptisols. This association is located mainly on strongly sloping to steep uplands, and is characterized as shallow, well-drained, and moderately permeable. Surface soil is a layer of gravelly sandy loam with a subsoil of gravelly loam to clay. These soils formed in material weathered from mica schist. The Tatum-Tallapoosa-Fruithurst soil association consists of Ultisols. This association is found on strongly sloping to steep uplands and plateau

edges, and is characterized as well-drained, moderately deep to shallow, and moderately permeable. Surface soil is a loam or gravelly silt loam with a subsoil of silt loam to clay loam. These soils formed in material weathered from slate. The entire area is characterized by hilly to extremely dissected surfaces with slopes ranging from 10 to 45 percent (Spivey, 1974).

Climate and weather

The climate and weather of the Talladega Upland ecoregion is determined primarily by its location in the middle latitude and is classified as humid continental to humid subtropical (Bailey, 1995). The climate is influenced by moist tropical air from the Gulf of Mexico (Predmore et al., 2007) and the drier mid-continental jet streams. Winds are most frequently from the south and are associated with the Gulf Stream; however, northerly winds are common during the winter months (Henderson, 2006). Wind speeds are generally mild throughout the year with occasional low pressure cells common during the summer months creating strong down-bursts, high speed straight line winds, and favorable conditions for tornados.

The mean annual precipitation ranges from 1,250 to 1,350 mm with small amounts falling as ice and snow during the winter months (Harlin et al., 1961; Craul, 1965). Precipitation is distributed evenly throughout most of the year with small peaks associated with summer thunderstorms (Varner et al., 2003a). The wettest month is March (Spivey, 1974) and the driest time occurs from August to October. Summer conditions are long-lasting, humid, and hot with mean daily maximum temperatures ranging from 26° C to 32° C and mean annual temperatures of 17° C. Winter conditions

are mild with minimal precipitation in the form of snow. Data from Southeast Regional Climate Center (1956-2007) reports freezing durations are generally less than 4 days (Maceina et al., 2000) and daily minimum temperatures range from 0° C to 5° C . There is little evidence to support significant effects on climate due to altitude or relief on the Talladega Upland ecoregion; however, in specific areas, microclimate variations can be significant and are influenced by slope, aspect, soil properties, water availability, and topographic relief.

Longleaf Pine Forest Communities

Life history

Understanding the life history characteristics of longleaf pine, including its distribution, biological diversity, and natural history, is essential when considering restoration of these ecosystems. Information is needed to understand the relationships between historical disturbance events (e.g., hurricanes, ice storms, logging, collection of naval stores, and fire) and their effects on species' abundance, distribution, and associated community structures. The term "life history" is generally defined as "an organism's lifetime pattern of growth rates, colonization, survival, reproduction, and ability to tolerate stress" and is commonly used by ecologists associating succession with competition and changing resources (Kimmins, 2004).

Longleaf pine still covers much of its historical geographic range, however is scattered throughout the range in small and often broken pockets (Landers et al., 1995). The high variability in climate, precipitation, soils, topography, and community associations throughout its range reflect the generalist characteristics of the species (Van

Lear et al., 2005; Macenia, 2000). Many endangered species, most notably the red-cockaded woodpecker (*Picoides borealis*) and gopher tortoise (*Gopherus polyphemus*), inhabit longleaf pine communities throughout the native range, and the onset of the Endangered Species Act brought attention to the role that longleaf plays as a keystone species (Brockway and Lewis, 1997).

The historical pre-EuroAmerican forest composition was fairly homogenous throughout the range, with small differences mainly occurring in the understory. Early explorers described the forests along the Coastal Plain as being almost entirely all longleaf pine savannas and woodlands with a rich herbaceous layer of wiregrass (*Aristida stricta*), Curtis dropseed (*Sporobolus curtissi*), numerous bluestems, and panic grass (*Panicum* spp.) (Brockway and Lewis, 1997). In the mid- and northern range of longleaf, the overstory composition was dominated by large longleaf pines with co-dominant loblolly pine (*Pinus taeda* L.) and shortleaf pine (*Pinus echinata* Mill.) and scrub oaks (*Quercus* spp.) in the understory. The herbaceous layer was dominated by a number of grasses (Poaceae), asters (Asteraceae), and legumes (Fabaceae) (Varner et al., 2003b).

Longleaf pine is similar to other southern pines in reproduction and growth. Like all species in the pine family, longleaf is monoecious and catkins and cones are produced yearly during the growing season before buds emerge. Catkins may begin forming in July, while cones are formed during a relatively short period of time in August (Boyer, 1990). Pollination takes place in late winter or early spring, but fertilization does not occur until the following spring when the cones are rapidly growing. Currently there is only one named southern pine hybrid that is naturally occurring, the Sonderegger pine

(*Pinus sandergeri* H.H. Chapm.). This hybrid is a cross between a longleaf and loblolly pine (Chapman, 1922).

Cone production usually begins when trees reach greater than 30 cm dbh (Platt et al., 1993). Seed production is yearly, with large seed crops being produced at 10 year intervals (Walker and Oswald, 2000). Longleaf cones can range in size from 15 to 25 cm depending on where they are located in the crown. The number of seeds per cone can vary annually from less than 15 seeds per cone to about 50 seeds per cone (Boyer, 1990). Assuming longleaf stands are comprised of dominant and co-dominant strata, peak seed production per hectare is reached at 6.9 to 9.2 m²/ha of basal area.

Longleaf pine germination differs from other southern pine species in that seeds have the ability to germinate soon after they are dispersed. Typically, germination occurs within 2 to 5 weeks after the seeds are released; though under optimum conditions seeds can germinate in as little as one week after reaching the ground (Boyer, 1990). Optimum conditions occur following a surface fire that creates bare soil and releases nutrients (Early, 2004). Rapid germination reduces the possibility of seed mortality due to predators, disease, drought, or freezing.

Longleaf pine is considered very intolerant of shade and, therefore, develops best under an open canopy, though it can also regenerate well under mature trees (Hardin et al., 2001; McCay, 2000). In the montane region, while longleaf occupies xeric south and west-facing slopes and ridges, it also occurs and reaches its full growth potential on well-drained sites with moderately deep soils. Longleaf pine typically occupies less favorable sites due to its ability to withstand a high frequency of disturbance and generally reduced competition on such sites.

One of the most distinguishing traits that longleaf has in relation to other southern pines is the epigeal grass-stage that occurs within the first years of development (Schopmeyer, 1974). During the initial development of a longleaf pine seedling, very little above-ground growth occurs. The grass-stage can last for 2 to 6 years (exceptionally, 20 or more years) while all starches and nutrients are directed underground for development of the root system (Early, 2004; Henderson, 2006). The longleaf pine seedling does not produce annual rings while in the grass-stage, and the epicotyl does not elongate rapidly as most other southern pines do during this time (Henderson, 2006; Pessin, 1934). This genetic trait stalls above ground growth for reduced levels of canopy competition where optimum growth can be achieved (Early, 2004). The negative aspect of this adaptation is that the seedling is highly susceptible to diseases, particularly brown-spot needle blight (*Scirrhia acicola*) and very intense burning fires (Crocker and Boyer, 1975).

Height and diameter growth increase rapidly after the first three to six years in longleaf seedlings. Height elongations of 0.6 to 0.9 m are typical in the early stages of growth during the first two to three years (Boyer, 1990); however, the seedlings are very susceptible to damage from fires. Once beyond this stage, at diameters greater than 5 cm, they are again fire resistant. When longleaf reach the dominant and co-dominant strata they grow very straight and naturally prune branches that are no longer required.

In a study investigating development of plantation longleaf pine stands, Boyer (1983) found that longleaf pine growth was highly correlated to the planting site. Obtaining early height growth relies heavily on the duration of the grass-stage. Longleaf seedlings in old fields and on sites that were mechanically treated exhibited fast seedling

growth, whereas longleaf in unprepared overcut sites were slow to transition from the grass-stage and exhibited slower overall growth.

Longleaf pine can reach heights of 24 to 30 m tall and diameters of up to 60 to 75 cm (Hardin et al., 2001), but within montane systems, longleaf generally reaches heights of 18 m and diameters of less than 45 cm. Growth declines rapidly as the tree reaches maturity. A combination of slow growth and low mortality rates can make these trees exceptionally long-lived. The expected life span can range from one to four centuries, with some individual trees approaching 500 years of age (Henderson, 2006; Platt et al., 1988). Although longleaf pine has the potential for a long life span, few ever reach their full biological potential due to a high frequency of disturbance events, including localized high speed winds, severe fires, lightning strikes, ice storms, timber harvesting, and beetle infestation (*Ips* spp.) (Brockway and Lewis, 1997; Henderson, 2006). These disturbance events can also severely affect stand dynamics and growth potential.

Cultural & land use history in the montane longleaf pine ecoregion

Currently, the longleaf pine forests are comprised of primarily uneven-aged, mixed species stands of pine-hardwoods on xeric to mesic sites. Though much of the region is still occupied by montane longleaf pine, travelers moving through the mountainous portions of the longleaf range prior to the 20th century may have described the forested component differently than we see today (Early, 2004).

Areas within the longleaf ecosystem have been inhabited since the early hunter-gathers of the Clovis and Late Paleo Period cultures (12,000-9,500 BP) (Bonnicksen, 2000; Carroll et al., 2002), though it was not until the Hypsithermal Period (7,500-5,000

BP) and the last 4,000 years that longleaf ecosystems spread and became dominant throughout the southeast (Van Lear et al., 2005; Brockway and Lewis, 1997). Since this period, the region has had a history of variable land use by Native Americans.

Historically, thirteen tribes dwelled in Alabama, including the Choctaw and Creek, as well as tribes migrating west such as the Cherokee and Chickasaw. Native Americans burned around their settlements to reduce fuels and protect from wildfires (Van Lear et al., 2005) and also to improve agricultural lands and hunting habitat.

Prior to EuroAmerican settlement, combinations in disturbance frequencies, distributions and site factors were much different under Native American management, which helped to promote a high biodiversity within the montane longleaf pine ecosystem (Van Lear et al., 2005). Shortly after the arrival of the first English explorers, ca. 1580, a great resource for the English Navy was seen within the vast forests of the southeast. The English reported to Sir Walter Raleigh in 1584 of having seen “trees which found supply the English Navy with enough tar and pitch to make our Queen the ruler of the seas” (Early, 2004).

The earliest EuroAmerican settlement in the Talladega Mountains began roughly about 1832 and is recognized as the period when the Native American tribe was removed from northeastern Alabama (Griffith, 1972; Shankman and Willis et al., 1995; Van Lear et al., 2005). The early settlers often relied highly on open-range grazing in addition to farming practices and hunting. These goals of the early settlers were very similar to the earlier inhabitants. Burning in the longleaf pine forests became common again. During the early part of the 1800s, as the southeast’s longleaf pine region was being settled by Euro-Americans, pioneers cleared land for the purposes of agriculture, livestock grazing,

and settlement construction. At that time, movement and trade throughout the Southeast was highly restricted to waterways and short journeys to nearby townships. Settlements were typically restricted to lowland areas where the soil was rich and useable waterways permitted the shipping of goods (Early, 2004; Frost, 1993).

One year after the founding of Jamestown, turpentine collectors began moving southeast to seek additional longleaf pine for naval stores. The term “naval stores” includes the tar, pitch, and spirits of turpentine and rosin collected from longleaf pine. These commodities were required to build and maintain wooden ships throughout the southeast and were essential in waterproofing ships, repairing ships at sea, and for stiffening “rigging” ropes that sailors would climb.

There were two ways to obtain tar and pitch in the early production of naval stores. The first consisted of a burning tar kiln, which was an earth-covered mound of dead pine wood that was often referred to as “lighter wood”. The second method of obtaining tar and pitch consisted of boxing living longleaf pine. This method was more destructive but produced products faster and sometimes more pure than the original kiln method. The boxing of a living tree consisted of a cut on two face sides (four sides in large trees) where a pan was placed to collect runoff from the scrapings that would remove the bark and cambium. The runoff was placed in a barrel and shipped off to a distillery. Few trees throughout the South escaped turpentine boxing (Frost, 2006).

Many of the actions of the turpentine practice did not kill trees directly; however, workers were often careless with their methods, creating indirect mortality of trees. Dippers would leak gum throughout the forest while transporting runoff to barrels and the chips running up the sides of the trees were often deeper than recommended. The

prescription for long-lived turpentine orchards (typically four years (Frost, 2006)) included raking away the needles. A greedy operator or one who was short on labor might choose to continue cutting new boxes as opposed to tending the current stock. The carelessness of many orchards proved to be devastating (Early, 2004). A healthy and unboxed longleaf would maintain an impressive resistance against a single or even numerous fires, but boxed trees would often fall victim to a single blaze enhanced by the poor management of nearby resin-soaked needles on the forest floor.

Settlements began expanding throughout the South in the 1800s and timber became a more heavily used resource. The majority of cutting was done by hand and logs were transported one at a time by horses, mules, and oxen. Water-powered sawmills became popular in the mid-1700s where local waterways transported logs down stream (Frost, 2006); however, timber was still only used for local construction until the 1850s. In the later half of the 1800s railroads expanded throughout the region, buying land on all sides of the lines and transporting timber all across the Southeast. The intensive logging phase from the late 19th to the early 20th century removed virtually all remaining virgin forests in the South. By the early 1900s, cutover longleaf pine sites were quickly being occupied by scrub oaks and species other than longleaf (Mohr, 1986).

The lumbering period from 1870 to 1930 likely had the most historically significant impact on longleaf pine abundance throughout the southeastern United States. By 1931, it had become apparent that longleaf was not reproducing (Early, 2004). The opportunity for regeneration on cutover lands was lessened due to mismanagement and lack of remaining seed source.

Annual burning of the cutover lands continued, but as a result of heavier fuel loadings and slash due to logging the fires were more intense. In most areas, the longleaf did not regenerate following a harvest. Extensive harvesting depleted seed sources, and where longleaf could regenerate it was quickly outcompeted by faster growing species or killed by fire because the seedlings were too small to withstand annual burning. Occasionally, seedlings that did become established in previously cut stands were destroyed by feral hogs (*Sus scrofa*) (Frost, 1993).

The onset of larger open-range pasturing and row crop farming gradually slowed the tradition of open-ranged burning in the northern portions of the longleaf ecosystems (Van Lear et al., 2005; Frost, 1993). It was common in the eastern areas of the longleaf pine region to see 1,000-1,600 head of cattle being drove through the Carolinas and Virginia. Early settlers used open-range pasturing, but also desired productive farmland that was achieved through burning practices. The need for ample vegetation in grazing areas for livestock often conflicted with ideals and practice of promoting fire for productive farmland (Early, 2004). In addition to conflicting ideals of burning between the northern and southern regions of the longleaf areas, it was often the case that entire forests would burn due to poorly managed turpentine practices.

Although fire previously promoted pine abundance throughout the South, the federal management philosophy, adopted in the early 1900s, supported fire exclusion from all forested lands. Very little information was available pertaining to fire and ecological needs of longleaf pine until the 1940's (Early, 2004). Following the removal of longleaf, it was unclear to what effect fire suppression had on reestablishment and the remaining populations. Fuel loadings had significantly increased following

reestablishment by hardwoods and other pine species, and fuel loadings were enough to hinder any regeneration of longleaf pine.

Studies have shown that frequent and intense fires favor communities that are highly adapted to the effects of fire, promote early succession species such as grasses and pines, and hinder the growth of hardwoods in later stages of succession (Lafon et al., 2005; Lafon and Grissino-Mayer, 2007; Varner et al., 2005). The pre-EuroAmerican settlement composition of the southern savannas, southern Appalachian Mountains and other fire-mediated communities promoted growth of grasses, such as wiregrass, bunchgrasses and bluestems in association with fire tolerant tree species such as longleaf pine and occasionally shortleaf pine (Frost, 1993; Varner et al., 2005).

The intensive study of longleaf regeneration in the mid-20th century was committed to reversing the century-long decline; however, these studies were unsuccessful at developing methods for repopulating a stand following a clear-cut. By the 1960s the failure to achieve successful regeneration of longleaf drove landowners to plant loblolly and slash pine (*Pinus elliottii* Engelm.). In the early 1950s, Tom Croker and Forest Service colleagues noticed natural regeneration of longleaf occurring at a plot that had been clear-cut in 1948 (Croker and Boyer, 1975). The success in regeneration was attributed to the previous year's 1947 bumper seed crop. This was close to what foresters had known as a shelterwood cut and gave hope for natural regeneration in longleaf pine forests.

The development of nursery techniques at Alexandria and Pineville, Louisiana in the 1970s and 1980s helped to overcome many of the problems associated with artificial regeneration of longleaf seedlings. The technique of growing seedlings held in

containers meant that more successful plantings could be made by hand or machine. Although the replanting efforts initiated by the federal government in the late 1900s were beneficial, land that had once been home to longleaf pine was now held by private sectors. Most private landowners had no desire to change their lands and private timber companies maintained the fast-growing loblolly pine plantations for lumber resources. Today, as may have been the condition following the lumber period, longleaf pine occurs on less than 1.3 million hectares (Figure 2.2) (Maceina et al., 2000). The lumbering period, coupled with the alteration of fire regimes, has considerably changed the landscape of the longleaf pine ecosystem (Pedmore et al., 2007). In the South, loblolly pine plantations and secondary growth mid-succession oak-pine forests have replaced longleaf pine dominated communities.

Fire is an essential tool in the management of ecosystems and can be used to reverse ecological succession. Today, of the 1.3 million hectares of remaining longleaf pine ecosystems, only about half are frequently burned (Outcalt, 2000), and the other half is experiencing severe alterations in ecosystem structure and composition (Varner et al., 2005). Historically, fire was a significant factor in the U.S. and promoted the development and ecological separation of many plant communities (Frost, 2000). The pre-EuroAmerican settlement fire regime was largely responsible for limiting the number of hardwoods and confining their distribution to areas of steep slopes, relatively poor drainage, or low elevations near streams and rivers (Predmore et al., 2007). Vegetative communities in the longleaf pine ecosystem were shaped by frequent fires that acted to remove species that were intolerant of such a disturbance regime and controlled the sizes and distribution of the less-fire adapted hardwoods (Van Lear et al., 2005).

Role of Fire & Forest Fuels

Fire behavior is characterized by intensity (during burning) and severity (post-burning). The term fire intensity is described as the rate of energy release, or rate of heat release, per unit length of fire front, and is calculated using the heat yield of fuel, weight of available fuel, and rate of spread of the fire front (Tangren, 1976). The term fire severity describes the immediate effects of fire on vegetation, litter, or soils; fires are ranked from low to high severity by the post-fire appearance of the resource of interest (Stanturf, 2008). These terms will be used throughout this study to identify fire effects based on behavioral changes in litter fuel loadings and the number of trees scarred.

Fire patterns & behavior

Fire intensity and severity is controlled by a number of factors including climate, topography, localized weather patterns, fuel type, and fuel loading. Fire can manipulate ecosystems across spatial scales (e.g., severity and consumption of burn material) and temporal scales (e.g., succession processes and changing regimes).

Climate is a prevalent factor in all fire regimes and has complex effects at multiple spatial and temporal scales. Climate controls much of a fire's predetermined effect through cycles of dry years and wet years associated with El Niño–Southern Oscillation (ENSO), Pacific Decadal Oscillation (PDO), North Atlantic Oscillation (NAO), and Atlantic Multidecadal Oscillation (AMO). In addition to large-scale climatic oscillations, smaller-scale and more localized microclimate factors (e.g., precipitation and mean minimum and maximum temperatures) can have long- and short-term impacts on the ability of plants to grow, decay, and regenerate (Jenkins, 2005). The Palmer Drought Severity Index

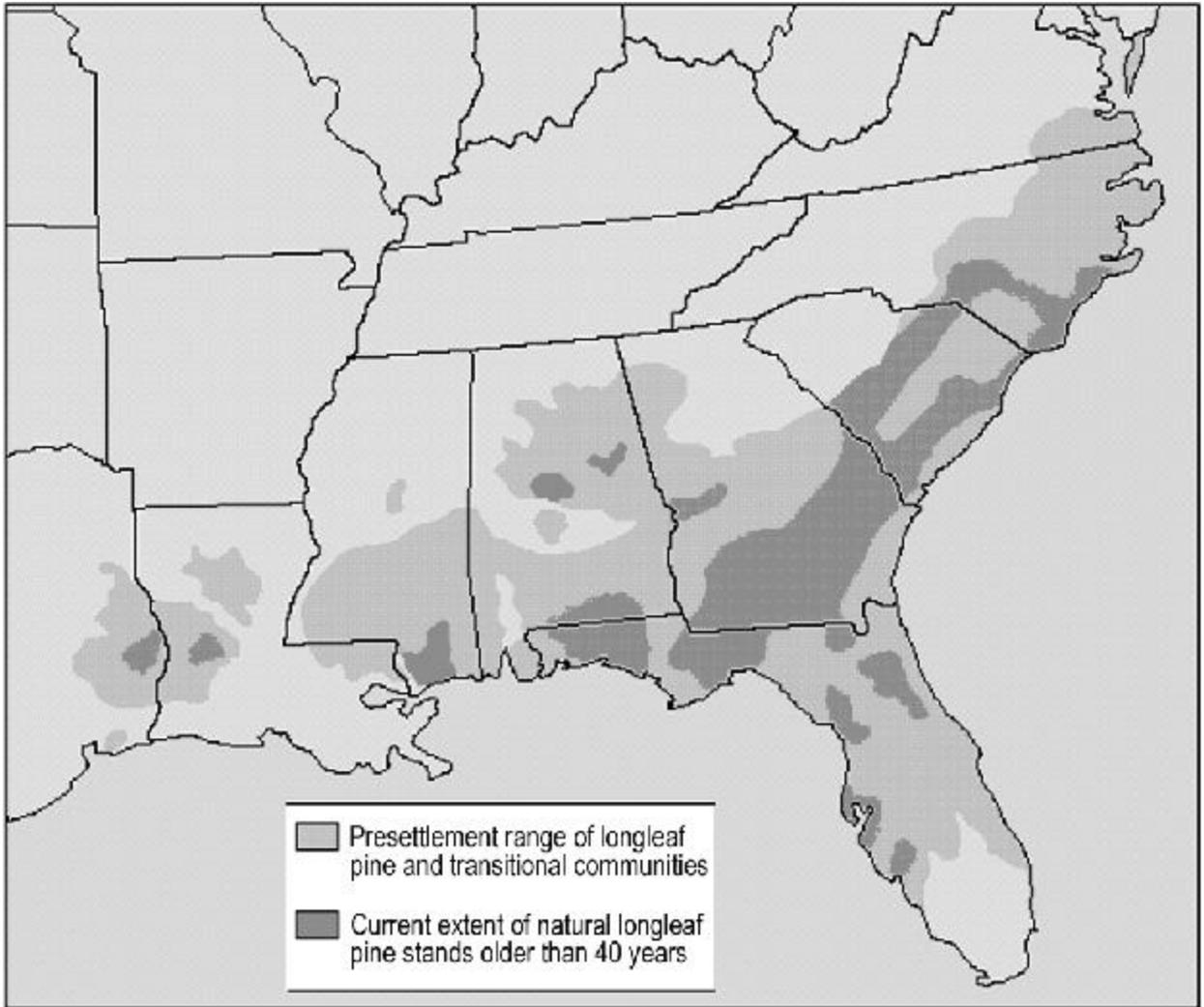


Figure 2.2. Historical and modern distribution of longleaf pine (White et al., 1998).

(PDSI), a combination of temperature and rainfall information in a formula to determine dryness (Cook et al., 1999), has been shown to correlate well with major fire events (Haywood et al., 2004) throughout the United States. This index has been incorporated in many maps to estimate fire risk.

Topography affects the fire environment through the modification of general weather patterns and creates localized weather conditions. Localized weather often influences the type of vegetation or fuels located throughout the landscape. Topography can alter the normal heat transfer process resulting in microclimates with localized wet and dry conditions. The rate at which a fire spreads is often a factor of topography. Topographic differences (e.g., elevations, slopes, aspects) cause variations in temperature and relative humidity and affects fuel type, loading, and moisture (Jenkins, 2005). South- and west-facing aspects will have higher fuel temperatures, lower humidity, more sparse vegetation, and drier fuels than north and east aspects. The decay process of plant matter is often slowed on south and southwest slopes, increasing the total amount of fuels and creating more intense fires.

Severely sloping terrain can directly affect intensity and direction of fires. Fires starting at the base of the slope can become larger due to the availability of fuel upslope. As a fire moves upslope and the slope increases, the fire will begin to move in a more prominent wedge, increasing the chances for spotting (NIFC, 1994).

Localized weather patterns can influence the behavior of fire more quickly than any other factor. Wind conditions must fall within the prescription plan when conducting a prescribed fire. Local winds are smaller-scale winds caused by local temperature differences and strongly influenced by terrain. Winds that are outside the prescription

range can hasten the drying of fuels (preheating) farther from the fire, increase intensity through an increased oxygen supply, increase the chance of spotting, or cause changes in fire direction due to wind shifts and convection (NIFC, 1994).

Fire research

Prescribed fire techniques used in the southern montane systems are typically justified based on fire history data (historical documents, charcoal sediments, and modern tree-rings) produced in the Coastal, Piedmont, or upper Appalachian systems (Bale et al., 2008). The use of fire history data from a different ecosystem influenced by a different climate can have significant negative effects on re-establishment, management, and site productivity (Lafon et al., 2005). While paleofire information is relatively abundant in the western U.S., this information is underrepresented in the eastern U.S. and even more so in the southeast.

Southeastern fire history information through pre-European settlement fire scar records is both easily obtainable and urgently needed for the successful restoration of longleaf ecosystems. Given the extensive information that tree-ring based fire histories can provide, there are some challenges associated with the method. These include:

- old wood, snags, and stumps are much less abundant in the East than in the West and are rapidly being lost due to recent burning or use as “lighter wood”;
- the fire scar record in live trees is rapidly being lost due to decay and mortality;
- multiple fire sites are often required for a complete assessment of the local fire history (e.g., capturing low severity fires);

- the season of fire is less synchronous with the season of tree growth in the eastern U.S. than in the West;
- many current fire histories from the Southeast are relatively short and do not predate Euro-American settlement.

The study of fire history through tree-rings has allowed researchers to provide information about historic fire frequencies, seasonality, and extent. This research has also linked fire events with rainfall patterns, temperatures, and anthropogenic activity (Henderson, 2006).

Fuel fluctuations, fire & litter relationships

Fuel is any organic material, living or dead, which can ignite and burn (NIFC, 1994). In this study, fuel will refer to a surface fuel that includes primarily needles or leaves, excluding the duff layer (Oe and Oa) and other woody biomass. Frequently burned pinelands have very little organic matter on the forest floor, but this condition has been altered by fire exclusion (Varner et al., 2005). The longleaf pine forests are perhaps among the most fire-dependent communities in the U.S. (Henderson, 2006) and are often referred to as being fire promoted (Frost, 2006).

Resinous needles and quick-drying woody debris from longleaf trees combine with diverse herbaceous vegetation to create an environment that allows fire to move quickly and burn very intensely. These fuels will dry completely within a few hours following a rain event (Varner, 2005). When long periods of fire-free intervals occur, a build up of fuels can become a major concern. Until recently, the effect on the intensity

and duration of burning events due to accumulations of litter and duff (partially decomposed plant matter) layers was unknown.

The ground cover within the longleaf community is unique. The structure created by the grasses, particularly bunchgrass and needlegrass, creates a perfect place for trapping longleaf needles beneath the canopy but above the soil surface. This creates a vertical structure for fuels where decomposition of needles is slowed and litter accumulates rapidly (Hendricks et al., 2002). The needles within this structure tend to burn more intensely due to increased oxygen availability, particularly during the summer months when evapotranspiration rates are high (Wahlenberg, 1946; Boring et al., 2004). When a volatile mixture of longleaf pine needles and grasses are combined with climatic drying events and high stocking levels, the entire forest stand can be detrimentally affected by fire. Where fire has been withheld for long periods, mature relic longleaf pines may be killed by smoldering in the accumulated organic matter at their bases (Van Lear et al., 2005). With a ground litter layer reaching almost one foot deep in some areas, the smoldering duff layer that lies below can often be more destructive to longleaf systems than a high severity and intense surface fire (Varner, 2005).

Longleaf, much like other pine species, are dependent on the nutrients that come from the partially decomposed material lying at their base. When long periods of fire-free intervals occur, roots begin to move upward to the surface to take better advantage of the vast amounts of nitrogen, phosphorous, and potassium that are leached out of the surface litter and into the soil (Boring et al., 2004; Varner, 2005; Varner et al., 2005). The reintroduction of fire can kill these roots that lie close to or within the duff layer.

This can possibly set back the growth and nutrient uptake of the tree and increase the rate of mortality.

Since fires usually burn within the surface layer, the horizontal continuity of these fine fuels is important. The horizontal distribution of needles and grasses influences where a fire will spread, how fast it will spread, and where the fire will travel given sufficient fuel (NIFC, 1994). Areas of non-continuous fuels are rare within the longleaf region, though they do occur in the mountainous locations where steep slopes, thin soils, and rocky terrain are typical. Within these mountain regions, partial burns are common, which can lead to pockets of unburned fuel accumulation; these pockets can lead to increased mortality when fire does once again come through the location.

In a study by Brockway and Lewis (1997), the benefits of periodic fire were outlined pertaining to community diversity, structure, and productivity in longleaf systems. The study examined influences of fire frequency on growth of plant biomass, litter accumulation, foliar cover, and herbaceous species richness, diversity, and evenness. They found that decay and mineralization rates were lowest in unburned sites and long fire intervals (>3yrs) and standing biomass of herbaceous understory plants was higher on all fire treated plots.

In another study conducted by Stambaugh and others (2006) in the Ozark Highlands hardwood region, the accumulation of leaf litter was overlain against varying lengths of fire intervals. They found that about 50 percent of the litter accumulation equilibrium is reached within 2 years, 75 percent is reached within 4 years, and equilibrium is reached after about 12 years. Using these findings they were able to model

the variable burning frequencies and predict litter loadings based on the desired management objectives as well as reconstruct the historic litter loading dynamics.

CHAPTER 3

METHODS

Description of Choccolocco and Brymer Mountain sites

Potential fire history study sites were identified based on topographic features, aerial photographs, and ground reconnaissance to indicate the likely presence of viable fire records (e.g., suitable tree species, exposed aspects, sandy soils, remote areas, old growth, sites exhibiting oak or pine mortality, and vegetation associated with target tree species). The effort focused on capturing the pre-EuroAmerican settlement fire record. Once a potential site had been identified, a physical survey determined whether it was used for the study based on the number of potential samples (remnants, dead snags, and mature live trees), the presence of fire scars, and evidence for sufficient temporal depth (e.g., long tree-ring series). The areas were exhaustively searched to find as many fire scarred trees and remnants as needed to obtain the longest and most complete record of fire events.

Two fire history sites lie within the Talladega Division - northern section of the Talladega National Forest, Cleburne County, Alabama (33°45' N, 85° 33' W). Located on the southwest portion of the Appalachian Mountains within the Blue Ridge region, Choccolocco Mountain is home to one of the last remaining old growth remnant longleaf stands in the montane ecoregion (Varner et al., 2003b). The two sites are located approximately 15 kilometers to the east of Choccolocco Mountain where few remnant longleaf have managed to survive. The two sites occur on both southeast and southwest aspects and extend to spur ridges that are capped with Weisner quartzite (Varner et al.,

2003a). Spatial sampling size was limited to an area of about 1 km² to minimize area-based effects and create a spatial scale pattern at which to view fire severity.

Choccolocco Mountain

Fire scarred stumps were located throughout the slopes within this area, but the majority of stumps that recorded a high number of scarring events were found near the tops and shoulders of the slopes and confluences of the ridges. These slopes provide the most protection from catastrophic events. Indications of the older age of the trees included flat tops and large upper branches of various longleaf individuals, signs that they have been around for many centuries. The area is known to have been previously used by a cluster of red-cockaded woodpeckers (RCW), a species known to inhabit old growth stands. The area had been actively managed for RCW use. Management within the area included periodic burning and thinning of hardwoods and non-desirable pine species to create a longleaf pine savanna with an abundant herbaceous layer to better facilitate the RCW needs. Intermediate and advanced regeneration within the stand was minimal, indicating the need for shorter fire intervals to remove competing vegetation.

Brymer Mountain

Located almost entirely on two ridges facing south to southeast, the majority of fire recorders were found in protected midslope positions. The encroaching hardwood line coming from the valley bottom played a major factor in locating stumps that were at higher elevations. Due to the increased moisture holding capacity of the lower slopes, very few stumps could be located farther downslope. Where stumps on the lower slope could be obtained, very few scarring events were evident and the stumps had decayed very rapidly, possibly decreasing the ability of fires to damage the cambial tissue.

Advanced regeneration of pine was scattered throughout the site in openings beneath canopy gaps formed by windthrow and localized burning hotspots. These patches of regeneration were dominated by loblolly, which was outcompeting the longleaf and other slower growing species. Ground vegetation was abundant creating a vertical structure that captured pine needles above the ground.

Fine forest floor fuel variability

Litter sampling areas were selected based on GIS raster data depicting prescribed burning dates from the Shoal Creek Ranger District's fire management office and were mapped using ArcGIS (ESRI, Inc.). These data layers helped to pinpoint the individual burn units located throughout the Talladega National Forest. Knowledge of when a fire was last present in a stand was vital information for the study's chronosequence approach. This information aids in determining the annual accumulation of litter since the last fire and also where total estimate of fuels are nearest to the maximum potential.

In addition to the availability fire records, collection sites were also selected based on vegetation and topographic criteria. Preferred sites were heavily pine-dominated (longleaf where possible), where drainages and other low-lying areas were avoided as these were not historically pine-dominated areas. Southern aspects were given priority overall as they traditionally contain the highest amount of pine-dominated overstory and tend to be unfavorable sites for hardwoods.

Field Procedures

Longleaf pine remnant identification

Successful determination of the species of a remnant fire-scarred pine can be a very difficult task. In order to eliminate or reduce the possibility of misidentifying a longleaf remnant, only areas where longleaf was once dominant or maintains current dominance were considered for collection of fire history data. One reason to avoid misidentification is the concern for misdating.

Longleaf pine remnants appear to have some characteristics which can be used to distinguish it from shortleaf and loblolly. The residence time (ability to persist within an ecosystem) of longleaf and shortleaf pine remnants is much higher than for loblolly, which tends to degrade within a few decades. The resin content of longleaf is higher, giving the annual latewood ring growth a darker appearance than shortleaf and loblolly. The final indicator lies within the latewood growth patterns. Viewed at microscopic levels, the radial growth within the earlywood of longleaf tends to maintain a relatively constant pattern, while the latewood growth has a higher degree of variability than that of shortleaf. This highly variable latewood growth pattern could prove to be a better predictor of late season climatic events.

Fire history sampling

An exhaustive search and examination of all live trees and remnant stumps within each of the study sites was conducted to identify samples with the most fire scars and longest time series possible. Opportunistic sampling was used to reduce the possibility of missing low intensity and low severity fires. The majority of samples collected were

taken from landscape positions that are highly sensitive with respect to fire scarring (Stambaugh and Guyette, 2006), such as upper slope positions, xeric aspects, and protected ridge and valley confluences. Longleaf pine is considered a fire adapted species (Varner et al., 2003b) as it has very thick bark, is able to survive multiple fire scarring events, and saplings are very resistant following 3 years of growth from the grass-stage.

Cross-sections were collected from forty stumps, natural remnants, and live trees from each site that showed evidence of recording multiple fire scarring events. Sample numbers and global positioning system (GPS) information were recorded for each remnant collected. The cutting of remnant stumps was done at or as close to ground level as possible, where low intensity fires had more opportunity to heat the cambial tissue and scar the tree (Henderson, 2006). Cutting the stumps as close to ground level as possible also gives the closest estimation for “true” pith. Live trees that recorded multiple fire scarring events were sampled using a wedged cutting technique that minimizes injury to the tree. In addition to the sampling of remnants, cores were taken from eight live trees within each site to aid in the development of a master tree-ring chronology (discussed in laboratory procedures), which was used for the crossdating of samples and fire scarring events.

Cores from live tree were selected from the oldest trees using criteria based on large upper branch formation and flat-topped crowns to estimate their age. Cores were taken using a 5.15 mm increment borer 1.3 meters above ground. Diameters and GPS locations were recorded for spatial data information.

Fine forest floor fuel sampling

Litter (< 1 hour fuels) was collected and depth-to-humus measurements were recorded within the fire history study areas and surrounding areas of the national forest to obtain multiple variants on localized fire intervals. Sampling of fuel loads began in August of 2008 and was completed the same month to ensure the estimates were not temporally different. Sampling was conducted using a chronosequence where prescribed fire years were used as timeline data for yearly variation on litter loadings. The chronosequence approach involved sampling throughout the entire forest based on the last known year of prescribed fire, where overstory and understory compositions were similar (e.g., longleaf dominated, fairly open midstory, and positive regeneration).

Litter plots were sampled using 0.5 m² quadrats and were located along 50 m transects that ran perpendicular to the ridge contour, either from ridgetop to footslope or vice versa. Aspect on the slope was kept constant, where possible, to avoid any changes in canopy composition that could be attributed to sun exposure. Although the canopy composition changed as the position on the slope changed, it generally remained pine-dominated. Two sub-sample quadrats were located at a random direction on each side of transects every 10 meters. The distance to each plot was deemed a sufficient distance to capture changes in stand basal area but short enough to permit multiple sampling points and not have an overstory canopy change due to changing slope position. “Zero plots” (areas with greater than 50% rock substrate or brush cover) were avoided. Plots randomly placed on a zero plot were relocated at a different location to obtain a representative sample of the surrounding area. In addition to zero plots, some subplot quadrats were located on the uphill and downhill sides of trees. These plots are important

to the study based on randomness (more needles will be found on the upslope portion of a tree than on the downslope) and to obtain a good estimate of needle weight.

Theoretically, these two plots balance out and provide information useful when depicting fire behavior (e.g., fires typically are more intense on the uphill side of trees producing the classic fire scar with the death of the cambial tissue due to the intense heat produced by these fires). Five plots were sampled along each transect. Each of these five plots were associated with two subplots, giving a true sample size of ten samples from each transect. The litter from each subplot was collected in one-gallon plastic zip bags and given a sample number that corresponded to the year of the last fire and transect number. Each burn year (time since fire (t)) was sampled twice where possible to increase the sample size and give added strength to the data set. Spatial variation in the data was achieved by locating and collecting these additional burn units in different stands throughout the national forest. This allows the data set to maintain independence spatially so that it may be used in a multiple regression analysis.

Raster data were summarized for collection years ranging from pre-1995 (the extent of the GIS data set) to the most current year (2008). The exception within this data set is found in 1999, as no burn units could be located that met the sampling criteria. Lacking data in the burn year of 1999 should not pose a problem in the analysis portion of these results as this data can be interpolated.

Information collected at each plot consisted of slope, aspect, number of trees (variable radius for BA), pertinent visual observations such as depth to humus or soil, overstory density, and groundcover density. Overstory and ground cover densities influence the amount of leaf litter that is captured within the vertical ground structure and play a key

role by decreasing the decay rate of longleaf pine litter. Increased ground cover (within open-canopy areas) can lead to more intense fires; fallen needles will not be close enough to the ground to allow moisture buildup and microbial decomposition to occur, but these fuels will be exposed to ample air movement, which can increase fire intensity.

Laboratory procedures & data analysis

Fire history

The development of long tree-ring chronologies was necessary to accurately date remnant samples collected at each of the fire history sites. The lack of dendrochronological work in the mountainous longleaf systems required the development of individual chronologies. In the laboratory, cross-sections surfaces were prepared through standard techniques of planing and sanding with progressively finer grit sandpaper (80 to 600 grit) to reveal cellular detail of annual rings and fire scars (Guyette et al., 2006; Orvis and Grissino-Mayer, 2002). Cores were glued and dried on wooden mounts and sanded.

Annual earlywood and latewood growth rings were measured twice from two radii of each remnant cross-section and core. Measurements were recorded using Measure J2X Java-based measurement software and a binocular microscope fixed to a 24 inch Velmex™ moving stage. Movement of the stage was measured by an electronic transducer with an accuracy of 0.01 mm. Annual ring width files were imported into COFECHA (Holmes, 1983) — computer software used for quality control of measurements and verification of dating accuracy. COFECHA checks the accuracy of dating by correlating successive time segments of each series. Each series was set at a

50-year segment length and lagged 25 years. This process overlaps each individual measurement with all other measurements until the highest correlations are found.

Since there were no previously published neighboring mountain longleaf pine chronologies, the creation of a site specific ring-width chronology was required for the accuracy of crossdating remnant samples. An ARSTAN tree-ring chronology was constructed for the Choccolocco Mountain site using Turbo ARSTAN software (Cook and Holmes, 1984; Cook, 2002). ARSTAN derives a standardized tree-ring chronology using a robust mean value function to reduce the effect of statistical outliers. Standardized chronologies are pre-whitened by using pooled auto-regressive models to form the auto-regressive standardized (i.e., ARSTAN) chronology (Cook, 1985; Stambaugh and Guyette, 2004).

A two step process was used to interactively detrend the tree-ring series (Holmes et al., 1986). First, to remove age related growth trends, a negative exponential growth curve or linear regression line was fitted to individual series. Second, to remove higher frequency variation in growth due to stand disturbances, flexible spline curves were interactively chosen. All tree-ring series were fit with a standard 32-year spline curve as no prior stand disturbance dates were known. Stabilization of the variance was analyzed using \bar{r} weighted stabilization method.

Ring-width series of each measurement were plotted and used for visual crossdating over previously known dated samples and cores (Stokes and Smiley, 1968) in addition to statistical procedures. Visual crossdating is perhaps the most crucial procedure in tree-ring analysis because, depending on species and location, a large

number of absent or false rings can and do occur and is important that the proper calendar year is placed to each annual ring (Fritts, 1976).

All remnant cross-sections were crossdated using previously dated samples, living tree cores with known dates, and by identification of extreme climatic variations which cause fluctuations in annual growth patterns. Each sample was dated using both latewood-only measurements and earlywood /latewood (full ring-width) measurements to determine the most accurate dating. Once dated, each sample was marked with pencil to signify decadal years enabling easier dating of injuries.

Fire scars were identified by the presence of cambial injury, callus tissue, traumatic resin canals, charcoal residue, and liquefaction. Fire scars were defined as wounds created when excessive heat or direct contact with a fire (scorching) causes death of the cambial tissue (Clark, 2003; Smith and Sutherland, 2001). The scars were dated to the first year of growth response in relation to the fire injury and to the season of injury where possible. FHX2 software (Grissino-Mayer, 2001) was used to generate summary statistics and to create individual tree and master fire chronologies for each site. Separate fire scar analyses were conducted for each site and included data for each individual sample. Mean fire intervals (MFI) were calculated with analysis beginning at the first year of tree-ring record and ending the year of collection for living trees, and refers to the mean number of years between fire scars (e.g., fire events) (Dieterich, 1980). FHX2 output analyses consisted of fire-scar dates, inner- and outer-ring dates, recorder fire years, and minimum and maximum number of scars per tree and site.

The percentages of trees scarred per site were calculated to create a spatial reference of the severity of each fire event (located on lower portions of master fire

chronologies). Due to longleaf's natural resistance to fire injury through its thick bark, many years of fire may be present in an area and only produce enough heat to injure or scar one individual tree. When both sample size and percentage of trees scarred are high, the fire event was of high severity and intensity.

Fine forest floor fuels

In the laboratory, 520 individual sample bags were weighed wet and then dried in ovens at 60°C for three to five days to obtain a constant oven-dried mass. While drying, one bag was randomly selected from each oven and weighed every day to monitor changes in mass characteristics. Collected litter included small amounts of bark, twigs, and pinecone scales. The amount of miscellaneous matter, other than leaf litter, was measured to be less than 3% of the total weight in each sample, thus, insignificant to the objectives of the project.

Data were compiled into an Excel spreadsheet and separated by number of years since the last known fire in the sampling area and transect number. Mass was recalculated from grams/ 0.25m² to tons/ acre. Percentage of total pine leaf mass versus hardwood leaf mass was estimated. The amount of pine-to-hardwood leaf mass was estimated to determine if any significant effects based on mass were identifiable with more or less hardwood mass to pine

Mean plot values of mass were calculated from the total number of individual plots located along transects (n=20). Subplots within the original sample plots (n=40) were added together to produce a more robust estimate of the true means located within the areas, then the average for each transect was calculated along with standard

deviations. The transect means were used to produce a litter accumulation curve over time.

Litter accumulation rates can be difficult to predict because of the high variability imposed by changes in species composition, vertical structure of vegetation, and faunal and microbial activity. The chronosequence method of sampling provides a look at long time scales and the changes that happen when fire is withheld from systems. The mass loss of litter as a function of time is generally expressed as an exponential decay model (Olsen, 1963; Stambaugh et al., 2006). The rate of accumulation of litter and the time required to reach maximum litter accumulation for the Talladega Upland longleaf pine forest was described using an exponential decay function (equation 1):

$$x_t = x_o - (x_o * e^{-kt}), \text{ (eq. 1)}$$

where:

x_t = amount of litter remaining after time t (years)

x_o = maximum quantity of litter (tons/acre)

e = exponential constant (approximately 2.71828)

k = estimated rate constant for litter decomposition (year^{-1})

t = time of accumulation (year)

Litter decomposition rate constants (k) were developed from two equations that incorporate very different variables. The first litter decomposition equation developed by Meentemeyer (1978) is a climatic based model. He presented a general equation for predicting average annual decomposition percentage from actual evapotranspiration (AET) and leaf lignin content (equation 2):

$$y_1 = -1.31369 + 0.05350 x_1 + 0.18473 x_2, \text{ (eq. 2)}$$

where:

y_1 = annual mass loss (%)

x_1 = annual AET (mm)

x_2 = AET millimeters/ Lignin (mm/%)

Based on Meentemeyer's equation, the annual mass loss percent can then be divided by 100 to obtain the average annual mass loss per year.

Actual evapotranspiration values for the study area were obtained through the Global Hydrologic Archive and Analysis System (GHAAS). The data were spatially oriented on a 0.5 x 0.5 degree grid that averaged annual AET estimates in millimeters per year. The AET values contain data used to compare water balance estimates where potential evapotranspiration was used as an estimator of actual evapotranspiration values over eleven U.S. watersheds (Vörösmarty et al., 1998). The grid data were loaded into ArcGIS (ESRI, Inc.) as a geographic information data set, where five grid points were taken and averaged together to determine the mean AET for the study area (Figure 3.1). The mean AET value estimated by Vörösmarty and others (1998) was consistent and within the range predicted by Lu and others (2003), using a different model that incorporates annual precipitation, watershed latitude, watershed elevation, and percentage of forest cover from 39 watersheds throughout the southeastern United States.

The mean AET value for the study area was incorporated with average litter lignin content percentages for longleaf pine. Lignin content percentages were used from

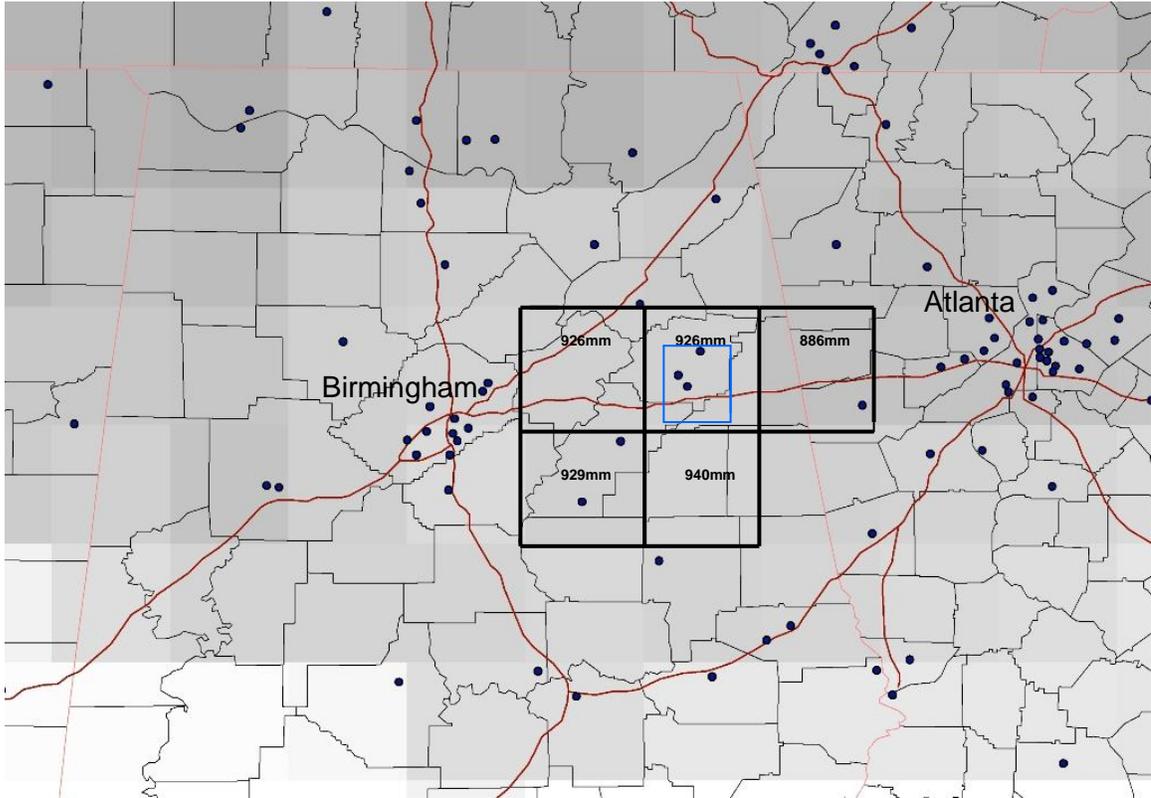


Figure 3.1. ESRI ArcMap data set from the U.S. Hydrologic Data (GHAAS) collection of the 0.5 x 0.5 degree grid of yearly actual evapotranspiration in Alabama. The five outlined grid boxes indicate the sampling area where the data were most relevant and a mean was derived.

previous published studies looking at tissue chemistry and carbon allocation of longleaf seedlings in the Southeast (Entry et al., 1998; Runion et al., 1999).

The second litter decomposition equation, developed by Olsen (1963), relies on actual measurements of litter mass. He presented a fractional equation where the annual production of litter is divided by the standing crop litter. This equation assumes a steady replacement of litter input (equation 3):

$$k = x/x_o, \text{ (eq. 3)}$$

where:

x = oven dried weight of mean mass annual litter production (tons/acre/year)

x_o = mean mass total litter accumulation at t_{\max} (>14) (tons/acre)

Olsen also found that a virtue of the simple exponential model is that the time required to reach half of the asymptotic level is the same time as that required for decomposition of half of the accumulated organic matter.

The two decomposition models presented above (equations 2 and 3) provided decay rates that were inserted into the exponential mass loss equation. Two distinct accumulation curves (equation 1) were then graphed against the mean litter loading values of the original sample plots to compare slope and asymptote. Identifying the accumulation model that best represents the overall litter build-up of the mountain longleaf system is beneficial to predicting future settings.

An additional decay constant was derived from averaging years 4 through 14 of time-since-fire. This constant was created to remove some of the stand basal area effects on the higher amount of litter accumulation toward the end of the sampling

chronosquence (equation 4). This decay constant from averaged time-since-fire years was then inserted into the exponential mass loss equation.

$$k = x/x_1, \text{ (eq. 4)}$$

where:

x = oven dried weight of mean mass annual litter production (tons/acre/year)

x_1 = mean mass litter accumulation of years 4-14 of time-since-fire (tons/acre)

In addition to using models that were derived from the exponential decay function and mean transect litter measurements, a logistic regression model was developed from the combination of all plots, transects, and years since fire. The data were separated by time since fire, transect number, and plot number in an Excel spreadsheet. The data were uploaded in SAS (SAS/STAT, 2002) software where it was then merged based on creating 20 separate blocks where the data was repeated. Litter data were transformed using a natural logarithmic transformation, where the procedure function was used to produce a stepwise model to determine the best fitting variables that could predict litter accumulation. Predictor variables were created on the assertion of having significant influences on yearly litter inputs. These predictor variables included time since fire (tsf), the square root of time since fire (sqrttsf), reciprocal of time since fire (rtsf), time since fire² (tsf²), and basal area (ba).

Ecological influences on yearly litter accumulation are difficult to determine and model. Factors such as stand basal area and stand structure can produce differences in yearly litter fall inputs. To determine stand-based effects on litter accumulation, measurements were taken from each year's time-since-fire and divided by the average

stand basal area for that year. Weighting each transect year by stand basal area for that year produces a value that reflects litter loading and the forest canopy at each year of time-since-fire.

Climate data

Divisional climate data (1895-2007) from the National Climate Data Center (NCDC, 2008), including monthly precipitation, monthly temperature, and monthly Palmer Drought Severity Index (PDSI) values, were used for analysis of climate response. Monthly data were averaged for yearly effects and graphed against yearly litter loading measurements. SAS was used to correlate climate data and mean litter measurements to analyze and choose the response variables. Although principal components regression is typically used to determine climate response functions, stepwise regression was used to keep monthly variables distinctively defined.

Because total litter accumulation is a combination of each year's input to the system, PDSI values should correspond with the litter input. PDSI values that were directly related to the post-fire year and all years associated with that fire year were achieved by adjusting the yearly PDSI values with averaged litter accumulation masses. To weight the PDSI values, it was important to know the contribution of litter for each year. PDSI values were multiplied by their respective percentage of contribution and added together to create a weighted (or "actual") drought index value with respect to time-since-fire.

CHAPTER 4

RESULTS

Tree-ring analysis

Tree-ring chronologies

Dendrochronological methods were used to create ring-width chronologies that were then used to date fire-scarring events at each site. Chronologies were developed for both the Choccolocco Mountain site and Brymer Mountain site. Because the two sites were spatially close, relatively, it was plausible to date one site with the other once one chronology had been created; however, for dating accuracy and independence, it was more logical to develop each based on their own data.

Remnant samples were cut near the root-stem interface to obtain as many growth rings and smaller fire events as possible. One of the problems associated with collecting samples from the root-stem interface is the unusual pattern within growth rings and ring structures. False rings (false spring), missing rings, and conjoining rings occur closer to the root-stem interface and when trees are young (Henderson, 2006).

Choccolocco Mountain

The Choccolocco Mountain chronology (Figure 4.1) was constructed using latewood-only ring-widths, which proved to maintain higher interseries correlations throughout all individual sample series. The chronology spanned a total of 460 calendar years (1547 to 2006 C.E.) and was based on both cross-sections and live tree cores. The oldest sample collected dated to 1547 C.E. and was also the longest tree-ring record (381

Choccolocco Mountain Latewood Standard Chronology

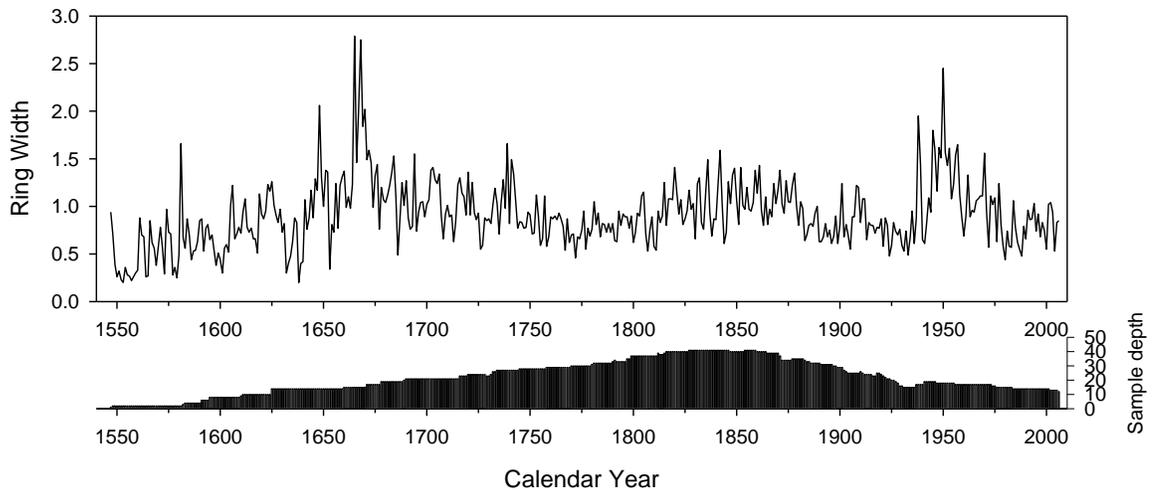


Figure 4.1. Standardized ring-width chronology of Choccolocco Mountain site comprised of 49 living and remnant cross-sections and 6 living tree cores. Each sample was interactively detrended using a negative exponential growth curve and then standardized to mean values to detect growth and climate signals and patterns. Mean and maximum number of rings on samples was 182 and 381, respectively. Total number of samples at each point in time is represented below the standardized chronology.

annual rings). The chronology included 55 dated samples, a total of 10,045 rings, and had a mean series intercorrelation of 0.506.

Absolute dating of remnant wood samples was done using six living tree cores that overlapped 10 remnant series. Once these samples had been given known calendar years to each growth ring it was possible to extend the chronology using 50 year segments lengths, overlapping the new series every 25 years, and comparing the correlations. This process enabled the dating of the oldest piece (pith date 1548 C.E.) within the Talladega National Forest and is the second longest age obtained for any known longleaf pine (Bale and Stambaugh, 2008). Dating problems suggested by COFECHA were plotted against the master chronology. No absent or missing rings were detected through the program.

Brymer Mountain

The Brymer Mountain site chronology (Figure 4.2) was also constructed using only latewood ring-width measurements. The chronology spanned 295 calendar years, from 1634 to 1928 C.E., and was created using only remnant cross-sections. The Brymer Mountain chronology included 35 dated samples, a total of 5,273 rings, and a mean series intercorrelation of 0.501.

The dating of the second site differed from the first due to a lack of cross-sections or cores taken from live trees. Individual ring-width measurements from Brymer Mountain samples were initially given an outer ring date of 1800 C.E. A “floating” chronology, not absolutely placed in time, was then created by running the individual series against one another in the COFECHA program. The floating chronology was then

Brymer Mountain Latewood Standard Chronology

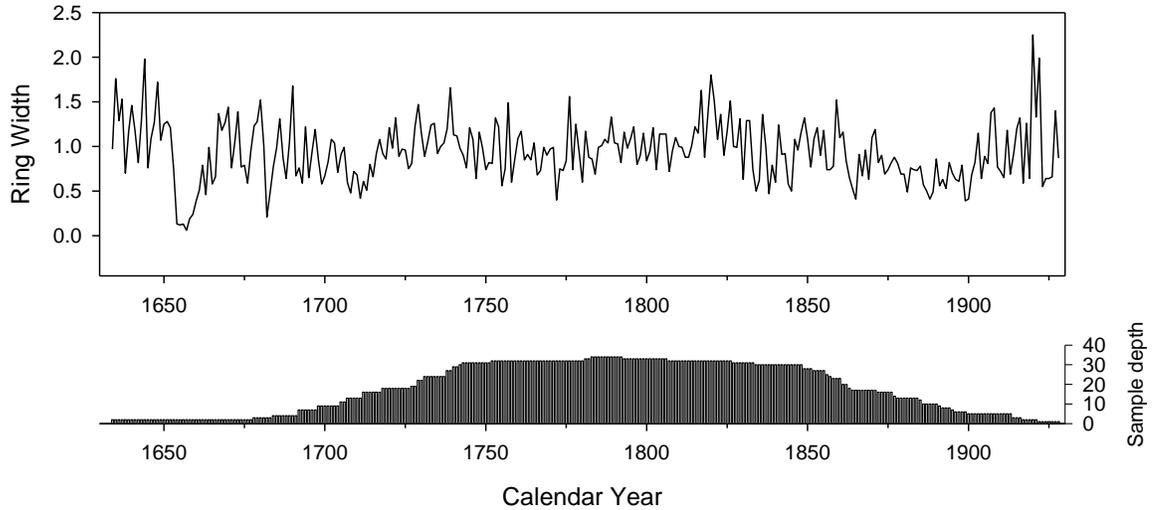


Figure 4.2. Standardized ring-width chronology of Brymer Mountain site based on 35 remnant cross-sections. Each sample was interactively detrended using a negative exponential growth curve and then standardized to mean values to detect growth and climate signals and patterns. Mean and maximum number of rings on samples was 150 and 211, respectively. Total number of samples at each point in time is represented below the standardized chronology.

given an absolute date by crossdating with the Choccolocco Mountain chronology.

Dating problems suggested by COFECHA were plotted against the master chronology to determine if the problem was true or false. No absent or missing rings were detected through the program.

Fire history chronologies

The sample number collected at each site was large enough that chronology dates and fire return intervals could be confirmed. Because the two sites were separated spatially and dated independently, it was possible to compare the two sites. Dates of fire scars at each site were reliable for most of the tree-ring record until the earlier portions of the fire record where the cohort of fire events lessens possibly due to sample size (Table 4.1).

The number of scars per sample ranged from a minimum of 1 to 29 scars, with an average between the two sites of 7.5 scars per sample (Table 4.2). Choccolocco Mountain

A composite fire chronology for Choccolocco Mountain site (Figure 4.3) was developed from 179 individual fire scars that were seasonally distinguishable (e.g., dormant, early, and late growing season). The 179 fire scars represented a total of 113 years with fire. The period of record ranged from 1547 to 2006 C.E. (460 yrs.), but is insignificant for fire frequency testing before 1653. Fire scar dates ranged from 1550 to 2001, and the percentage of trees scarred during fire years ranged from 5 to 100.

Fire frequency for the pre-EuroAmerican settlement period from 1653 to 1831 had a mean fire interval (MFI) of 3.2 years. Following EuroAmerican settlement, fire frequency increased to 2.5 years from about 1832-1940 (Table 4.3). Some fire years

Table 4.1. Periods of fire history reliability.

Site	Beginning Year	Ending Year	First Fire Year	Last Fire Year	Date of Reliability
Chocolocco	1547	2005	1550	2001	1653-1940
Brymer	1550	1940	1660	1934	1660-1940

Table 4.2. Sample statistics for fire history sites.

Site	Total number of samples	Total number of fire years	Scars per tree		
			Minimum	Maximum	Average
Chocolocco	26	113	1	29	6.9
Brymer	24	106	1	28	8.0
Total	50	219	1	29	7.5

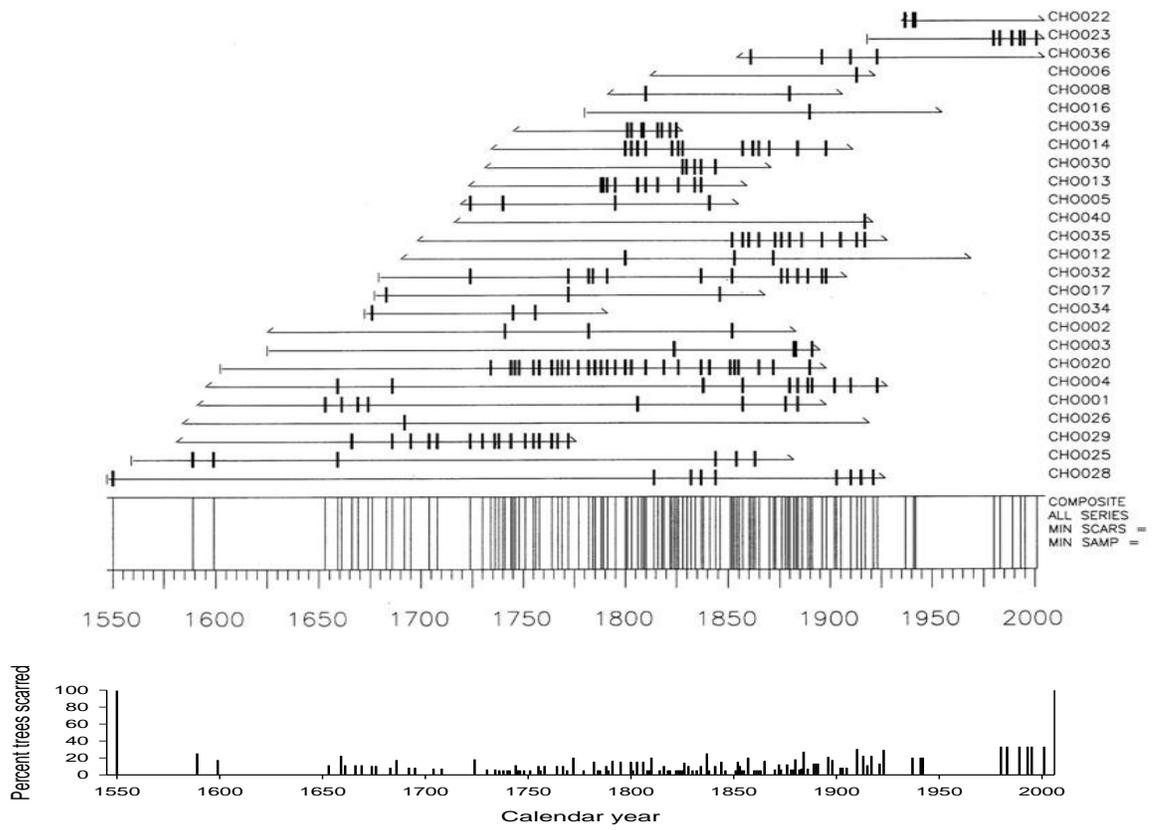


Figure 4.3. Composite fire scar chronology and fire scar dates of individual trees at Choccolocco Mountain, Talladega N.F. Calendar years are shown on the x-axis. Horizontal lines represent the period of record for each sample tree; bold vertical bars represent the year of a fire scar. Pith dates (year of tree regeneration) are represented by short, thin vertical lines at the left end of the horizontal line, while inside ring dates (first tree ring, pith absent) are represented by diagonal lines. Percentage of trees scarred is shown below the composite fire chronology and represents actual calendar years.

Table 4.3. Summary of fire intervals and percentage of trees scarred of pre- and post EuroAmerican settlement at Choccolocco Mountain.

Choccolocco Mtn.	Time Period	
	Pre-EuroAmerican 1653-1831	EuroAmerican 1832-1940
Mean Fire interval (years)	3.2	2.5
Mean Percent trees scarred	8.9	12.1
Number of years with fire	57	43
Percent of years with fire	32	39.8

during this period had high percentages of trees scarred, perhaps indicating a more severe fire regime than pre-1830s and post-1940s. The fire frequency at the site shifted again in the mid-20th century, becoming less frequent due in part to the advent of controlled burning policies in 1930s (Van Wagtendonk, 2007). During this period the MFI was 7.5 years, and includes the most recent time period where prescribed burning once again influenced the landscape.

Exceptional recorder trees (trees that have the ability to witness many fire scarring events as seen in Figure 4.4), such as CHO020, were located within some of the most protected portions of the site. These areas allowed enough fuel accumulation to carry a fire but not injure the tree past a recovery point. The combination of low intensity fire and protection allows the tree to record many centuries of fire events, because while the tree's wound is still open it remains more susceptible to recurring events, in the rarest cases of annual burning (Figure 4.5). These trees provide vital information about fire frequency and the climate regimes that might have influenced them.

The majority of scars occurred at the beginning of a growth ring, indicating that fires occurred most often during the dormant season (Figure 4.6). Dormant season fire scars constituted 97.2 percent of all fire events throughout the length of the fire chronology. Only 2.8 percent of fire scars were located in the middle earlywood to latewood portion, and only 0.6 percent of all fire scars were not seasonally distinguishable.



Figure 4.4. Annually scarred longleaf pine remnant on protected slope depicting classic fire-scarred “cat-face.”



Figure 4.5. Partial cross-section of CHO020 depicting annual and biannual fire scarring events.

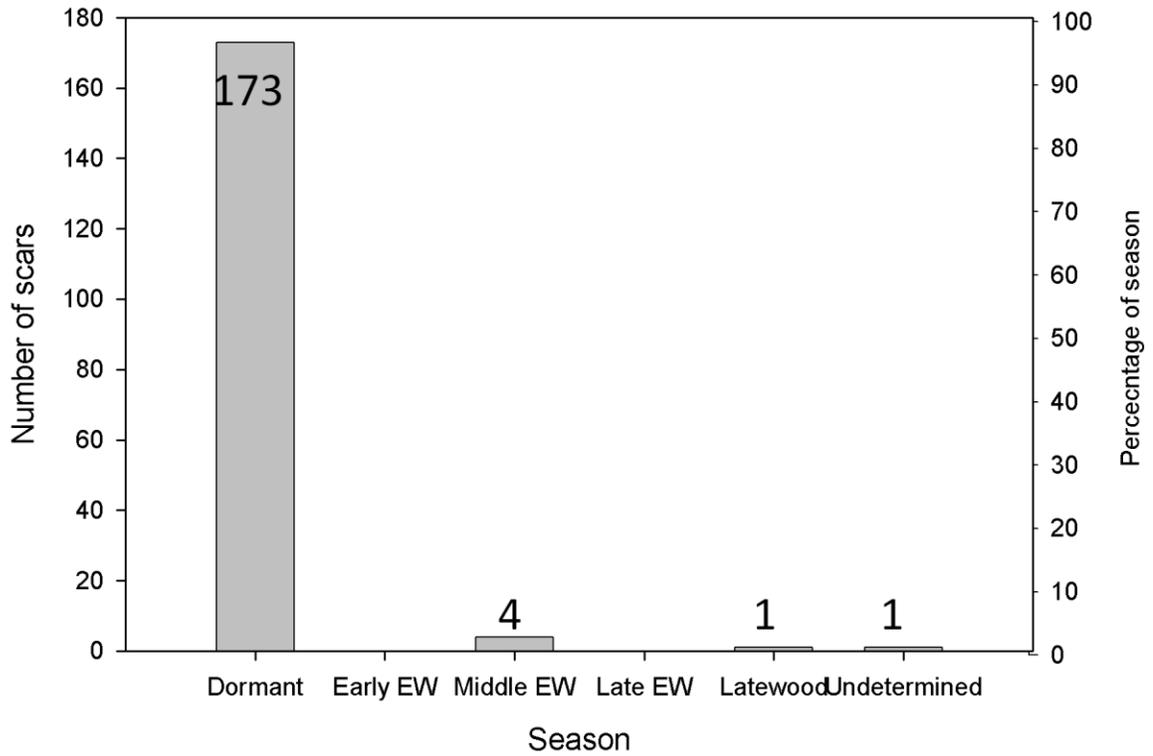


Figure 4.6. Summary of seasonal fire scarring events at Choccolocco Mountain. Number of fire scars per season is indicated above each bar. Abbreviation “EW” represents earlywood ring growth.

Brymer Mountain

A composite fire chronology for the Brymer Mountain site (Figure 4.7) was developed from 193 individual fire scars that were seasonally distinguishable (e.g., dormant, early, and late growing season). The 193 fire scars represented a total of 106 years with fire. The period of record ranged from 1550 to 1940 A.D. (391 yrs.), but is insignificant for fire frequency testing before 1660. Fire scar dates ranged from 1660 to 1934, and the percentage of trees scarred during fire years ranged from 4 to 100.

Fire frequency for the pre-EuroAmerican settlement period from 1660 to 1831 had a mean fire interval (MFI) of 2.7 years. Following EuroAmerican settlement, fire frequency increased to 2.6 years from about 1832-1940 (Table 4.4). Some fire years during this period had a high percentage of trees scarred, perhaps indicating more severe fire. Information for pre-1660 and post-1940 is incomplete due to lack of samples covering this time period.

Much like the Choccolocco Mountain site, the majority of scarring events occurred before the beginning of a growth ring, indicating that fires occurred most often during the dormant season (Figure 4.8). Dormant season scars constituted 92.4 percent of all fire events throughout the length of the fire chronology. Only 7.1 percent of scarring events were located in the middle earlywood to latewood portion, and only 0.5 percent of all fire scars were not seasonally distinguishable.

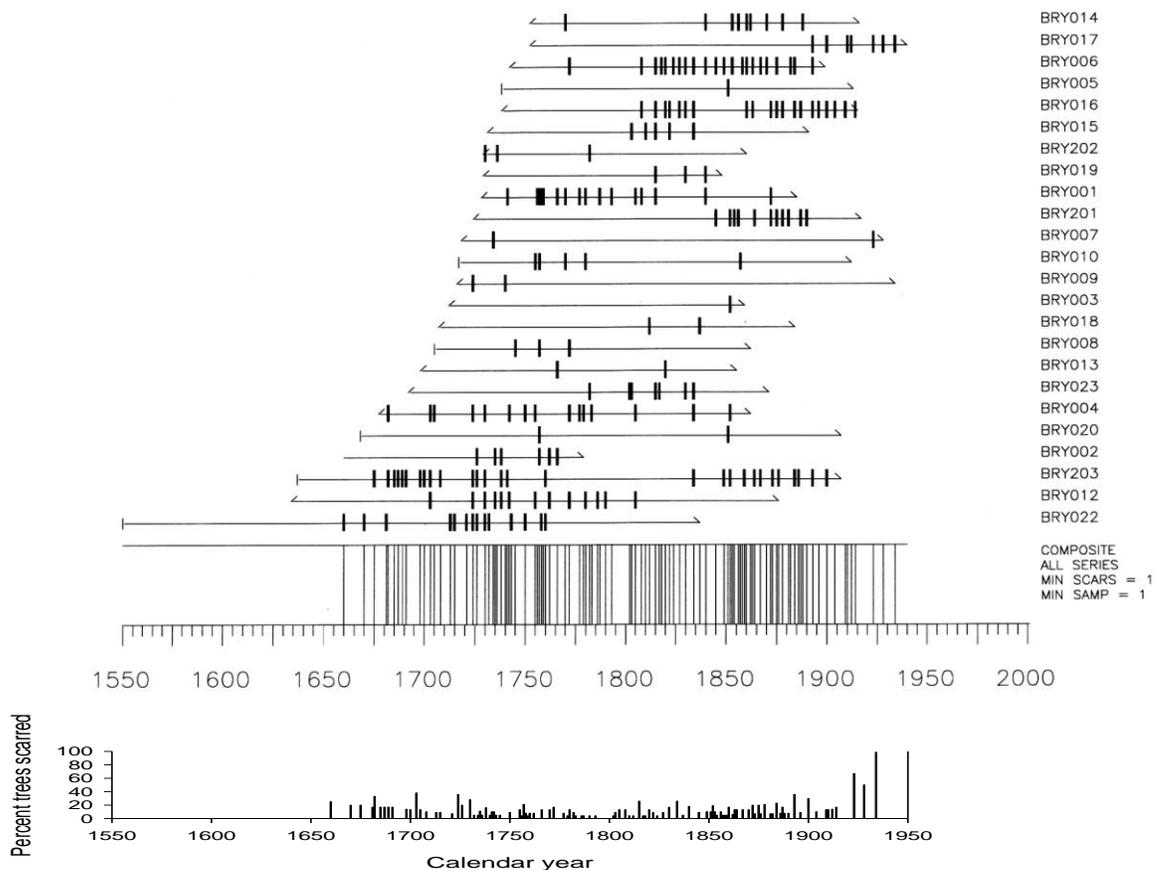


Figure 4.7. Composite fire scar chronology and fire scar dates of individual trees at Brymer Mountain, Talladega N.F. Calendar years are shown on the x-axis. Horizontal lines represent the period of record for each sample tree; bold vertical bars represent the year of a fire scar. Pith dates (year of tree regeneration) are represented by short, thin vertical lines at the left end of the horizontal line, while inside ring dates (first tree ring, pith absent) are represented by diagonal lines. Percentage of trees scarred is shown below the composite fire chronology and represents actual calendar years.

Table 4.4 Summary of fire intervals and percentage of trees scarred of pre- and post Euro-American settlement at Brymer Mountain.

Brymer Mtn.	Time Period	
	Pre-EuroAmerican 1660-1831	EuroAmerican 1832-1940
Mean Fire interval (years)	2.7	2.6
Mean Percent trees scarred	11.9	17.2
Years with fire	63	41
Percent of years with fire	36.8	38.0

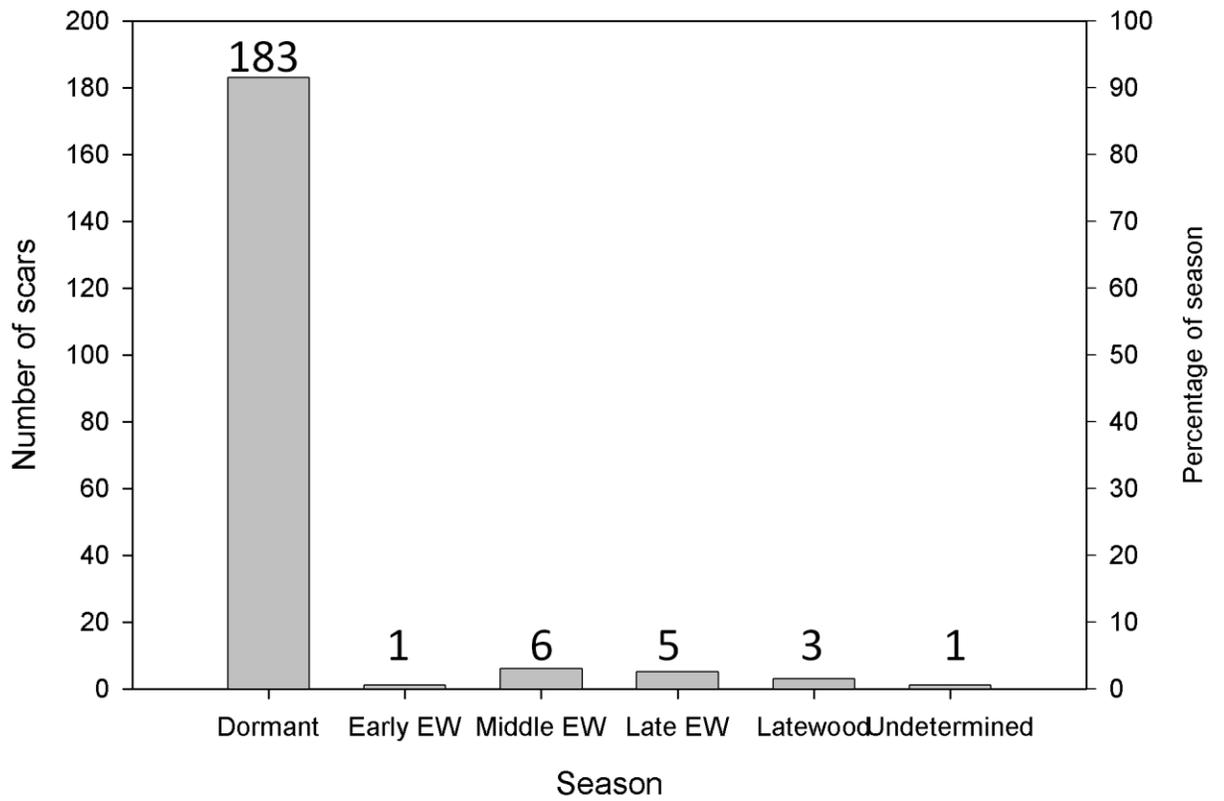


Figure 4.8. Summary of seasonal fire scarring events at Brymer Mountain. Number of fire scars per season is indicated above each bar. Abbreviation “EW” represents earlywood ring growth.

Forest litter analysis

Fine forest floor fuels

The chronosequence approach to sampling litter throughout the forest based on timing since the last fire produced annual litter loading estimates with respect to fire frequency for 14 consecutive years, with the exception of year 1999 for which no burn units could be found. As time-since-fire increases, litter mass begins to achieve a stabilization point due to increased effects of decomposition. Two commonly used decomposition rates (k) were used in the development of litter accumulation equations to determine at what point the maximum point of litter accumulation occurs. These two decomposition rates were developed from mass loss equations by Meentemeyer (1978) and Olsen (1963). Each of the decay rates were inserted into the temporal litter variability equation and compared to actual measurements taken from the forest floor.

A Litter loading and decay relationship was developed from Olsen's equation (eq. 3 in chapter 3) and yielded $k = 0.44 \text{ yr}^{-1}$. This rate was based on the ratio of mean mass of annual litter production (time = 1 year) to the mean mass of standing crop litter (time = >14 years). The mean mass of annual litter production was 2.33 tons/acre ($n = 20$, s.d. = 0.79) with minimum and maximum production of 0.83 and 4.15 tons/acre, respectively. The mean standing crop of litter was 5.32 tons/acre ($n = 20$, s.d. = 1.91) with minimum and maximum totals of 2.92 and 10.34 tons/acre, respectively.

An empirical litter relationship was also developed from Meentemeyer's equation (eq. 2 in chapter 3) and yielded $k = 0.55 \text{ yr}^{-1}$. This rate was based on average lignin concentrations for longleaf pine and actual evapotranspiration (AET) values. Actual

evapotranspiration values were first averaged through 5 different degree points and resulted in an averaged value of 921 mm/yr; however, the single value of 926 mm/yr, that was closest representative to the site, was used for a more accurate measurement of AET.

Lignin concentrations for Meentemeyer's equation were obtained through two previous publications on longleaf pine tissue chemistry. Concentration values of lignin varied from 21 percent (Runion et al., 1999) to 26.8 percent (Entry et al., 1998) between the two published studies due to treatment effects under elevated CO₂ and nitrogen fertilization conditions. Due to these varying lignin percent concentrations, decomposition values based on both the lowest and highest concentrations were calculated. Calculated decay rates were 0.56 yr⁻¹ for 21 percent lignin concentrations and 0.54 yr⁻¹ for 26.8 percent lignin concentrations. The difference between the two decay constants in relation to lignin concentration percents were insignificant; however, there were two different decay constants derived, therefore they were averaged to create a single climate driven decay constant to apply to the litter accumulation model.

The temporal litter variability model for the Talladega National Forest was described using an exponential decay equation and is presented in terms of percent accumulation and mass accumulation. The decay rate of 0.44 yr⁻¹, developed from Olsen's (1963) equation, was inserted in the temporal litter variability model (equation 5 and 6) to create accumulation curves based on actual measurements:

$$\text{Percent accumulation} = 100 - (100e^{-0.44t}), \quad (\text{eq. 5})$$

$$\text{Mass accumulation} = 5.32 - (5.32e^{-0.44t}), \quad (\text{eq. 6})$$

where t is the number of years of litter accumulation.

The decay rate of 0.55 yr^{-1} , developed from Meentemeyer's (1978) equation, was inserted in the temporal litter variability model (equation 7 and 8) to create accumulation curves based on climatic factors:

$$\text{Percent accumulation} = 100 - (100e^{-0.55t}), \quad (\text{eq. 7})$$

$$\text{Mass accumulation} = 5.32 - (5.32e^{-0.55t}), \quad (\text{eq. 8})$$

where t is the number of years of litter accumulation.

The decay rates (previously described) used in the temporal litter variability model equation are based on maximum accumulation values of year 14 of time-since-fire. These modeled decay rates estimate that litter accumulates to 25 percent, 50 percent, and 75 percent of maximum accumulation at approximately 1 year, 2 years, and 3 years, respectively. The difference between the two models is relevant at the equilibrium accumulation (99 percent). In the model derived from Olsen's decay constant of actual litter loading measurements, equilibrium is reached at approximately 10 years; however in the model derived from Meentemeyer's decay constant from climatic based variables, equilibrium is reached two years faster at approximately 8 years.

A decay rate of 0.55 yr^{-1} , that was created from the average of the last 11 years of time-since fire, was additionally inserted into the temporal litter variability model (equation 9 and 10) to create an accumulation curve that was based on the averaged litter mass of the last 11 years of time-since-fire:

$$\text{Percent accumulation} = 100 - (100e^{-0.55t}), \quad (\text{eq. 9})$$

$$\text{Mass accumulation} = 4.25 - (4.25e^{-0.55t}), \quad (\text{eq. 10})$$

where t is the number of years of litter accumulation.

Graphing of the modeled decay rates in addition to actual litter measurements (Figure 4.9 and Figure 4.10) helped to determine the decay rate that was best suited as a representative of the forest accumulation when related to decay.

The exponential growth-to-maximum (EGTM) model was developed using data from all plots and transects was correlated highest with a single variable. The variable reciprocal of time since fire (rtsf) provided an R^2 of 0.40 and produced a model equation with an intercept and constant associated to rtsf with 258 degrees of freedom (equation 11).

$$\text{accu} = e^{(1.34 - 0.132*(1/\text{tsf}))}, \quad (\text{eq.11})$$

where:

accu = estimated litter accumulation at tsf (tons/year)

e = exponential constant (approximately 2.71828)

tsf = time since fire (years)

The EGTM model assumes that directly after a fire there remains approximately one ton per acre of litter will remain. The equation estimates (Figure 4.11) that litter accumulates to 88 percent of maximum accumulation at approximately 1 year and 94 percent of maximum accumulation at approximately 2 years. The estimated equilibrium accumulation (99 percent) is achieved approximately at 7 years.

The initial view of basal area and individual plot litter loading measurements appeared to show a linear correlation (Figure 4.12). Weighting each transect year by stand basal area for the same year produced values that reflect litter loading and the forest

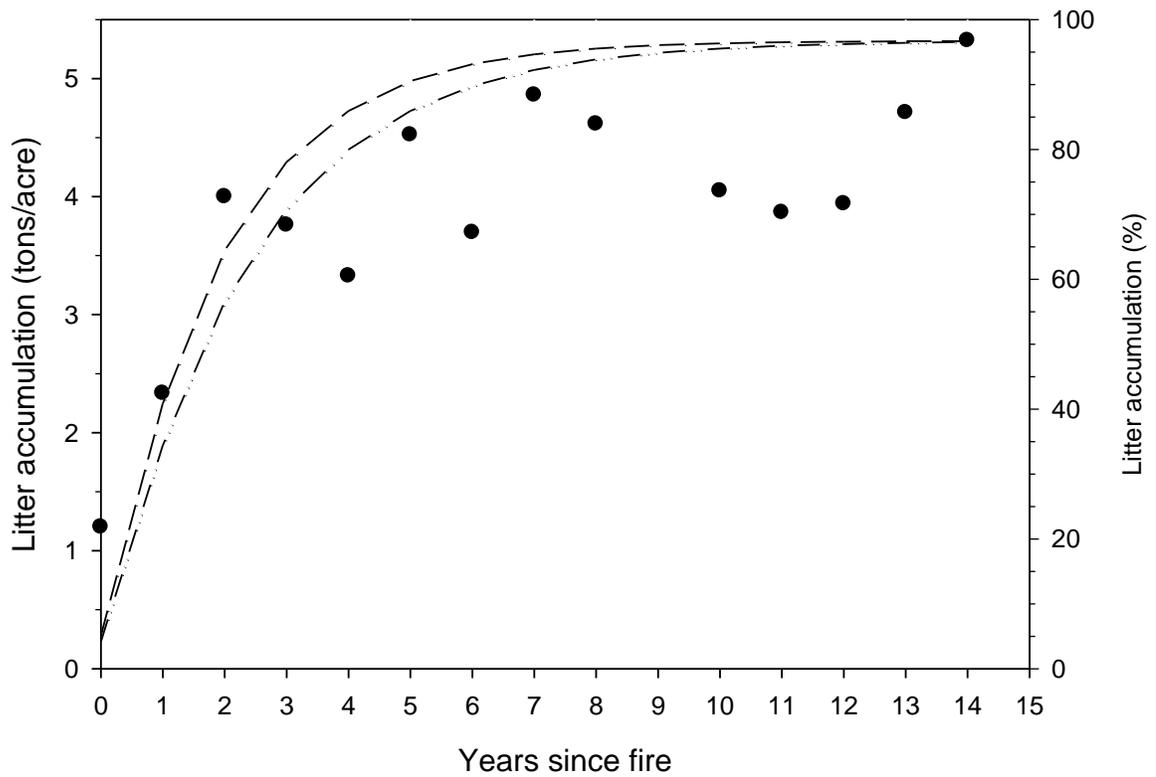


Figure 4.9. Summary plot depicting both empirical litter relationships developed from Olsen (dot dashed line) and Meentemeyer (dashed line) decay constants. Averaged yearly litter measurements are indicated by the bold circles and represent combined transects.

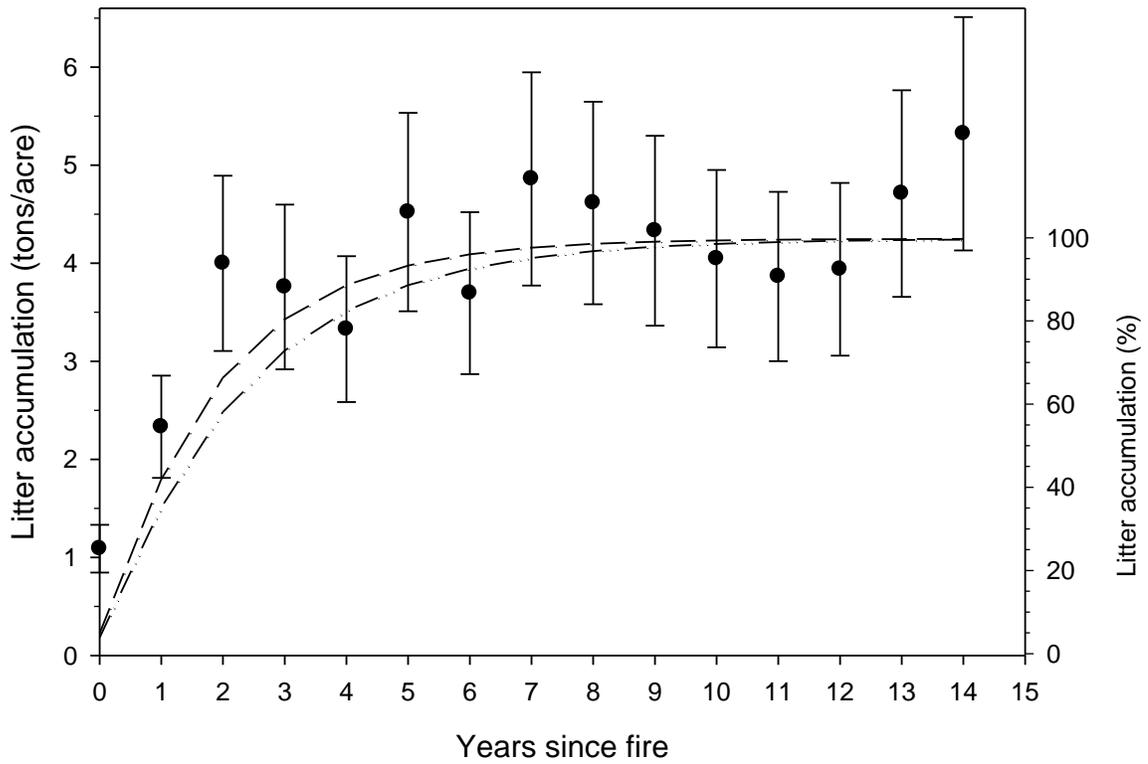


Figure 4.10. Summary plot depicting both empirical litter relationships developed from Olsen (dot dashed line) and Meentemeyer (dashed line) decay constants using averaged litter mass from years-since-fire of 4 through 14 . Averaged yearly litter measurements are indicated by the bold circles and represent combined transects.

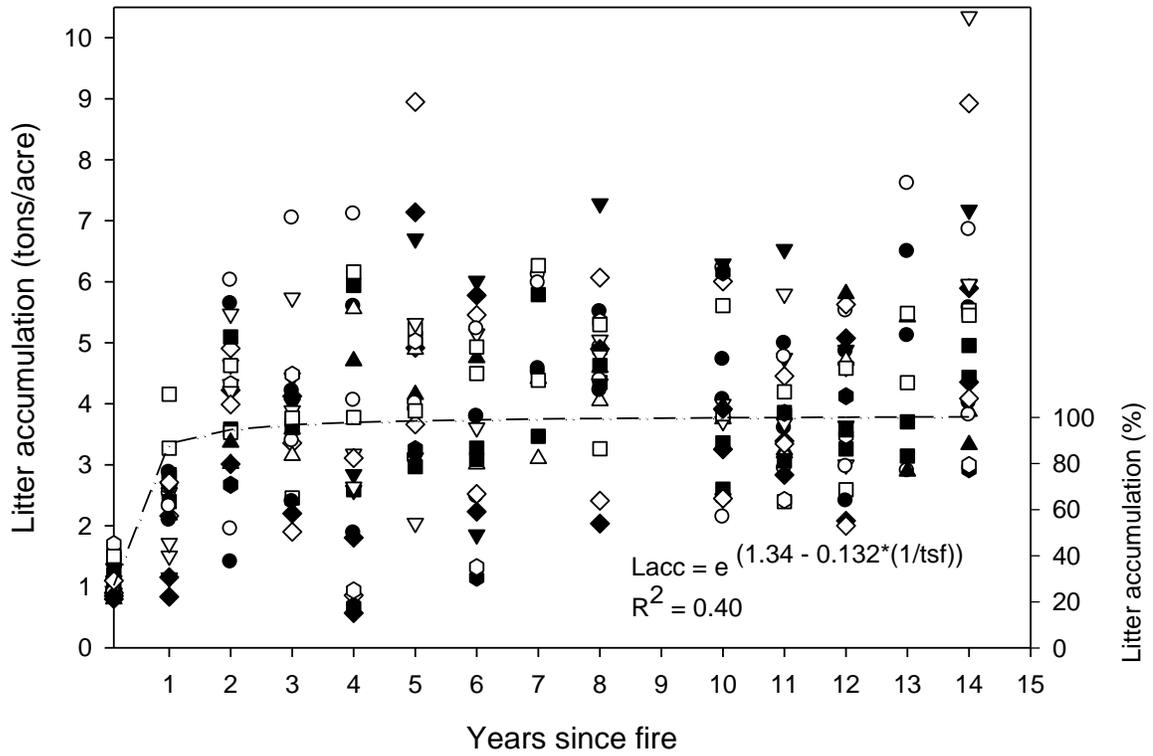


Figure 4.11. Plot illustrating litter accumulation measurements for all plots and transects with EGTM model (dot dash line).

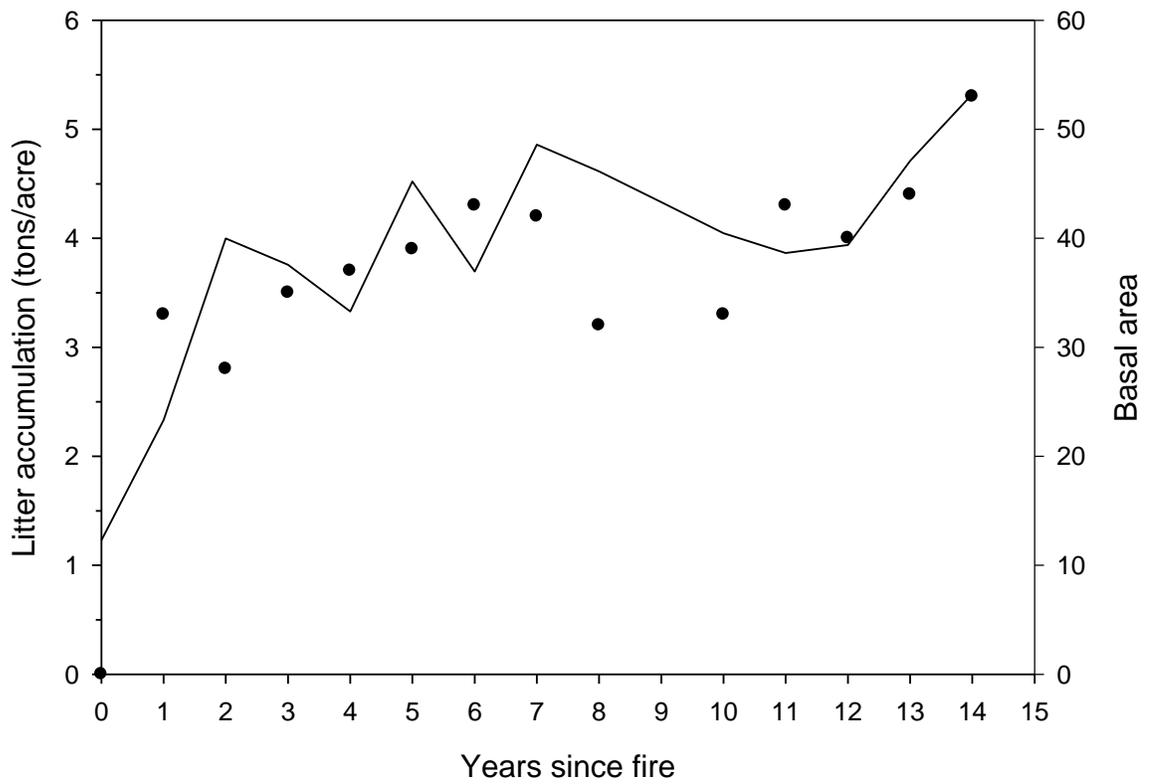


Figure 4.12. Litter accumulation measurements (bold circles) with average basal area per year (solid line).

canopy at each year of time since fire (Figure 4.13). During the initial litter accumulation stages (time-since-fire < 6 years; $t < 6$) a positive relationship exists between averaged litter weight and stand basal area. During the mid-portion of the chronosequence ($t > 6 < 11$), there is a drop in the measured litter loadings reflecting changes associated with basal area. As basal area increases through the later portion of the times series, litter loading begins to increase as it approaches the asymptotic level displayed in the modeled litter accumulation function.

Fuel, fire, & climate

The litter accumulation function showed important differences in litter accumulation when compared with varying burning frequencies (Figure 4.14). For example, an annual burning frequency allows a maximum of 35 percent of the total litter to accumulate. A burning frequency of two years allows a maximum of 58 percent of the total litter to accumulate, a burning frequency of five years allows a maximum of 89 percent of the total litter to accumulate, and a burning frequency of 10 years allows a maximum of 99 percent of the total litter to accumulate. In terms of litter loading, the difference between annual and 5-year burning frequency is more than two times greater than the difference between 5-year and 10-year burning frequencies. This relationship assumes total consumption of litter material at the point of a fire event and was based on values determined through Olsen's decay rate.

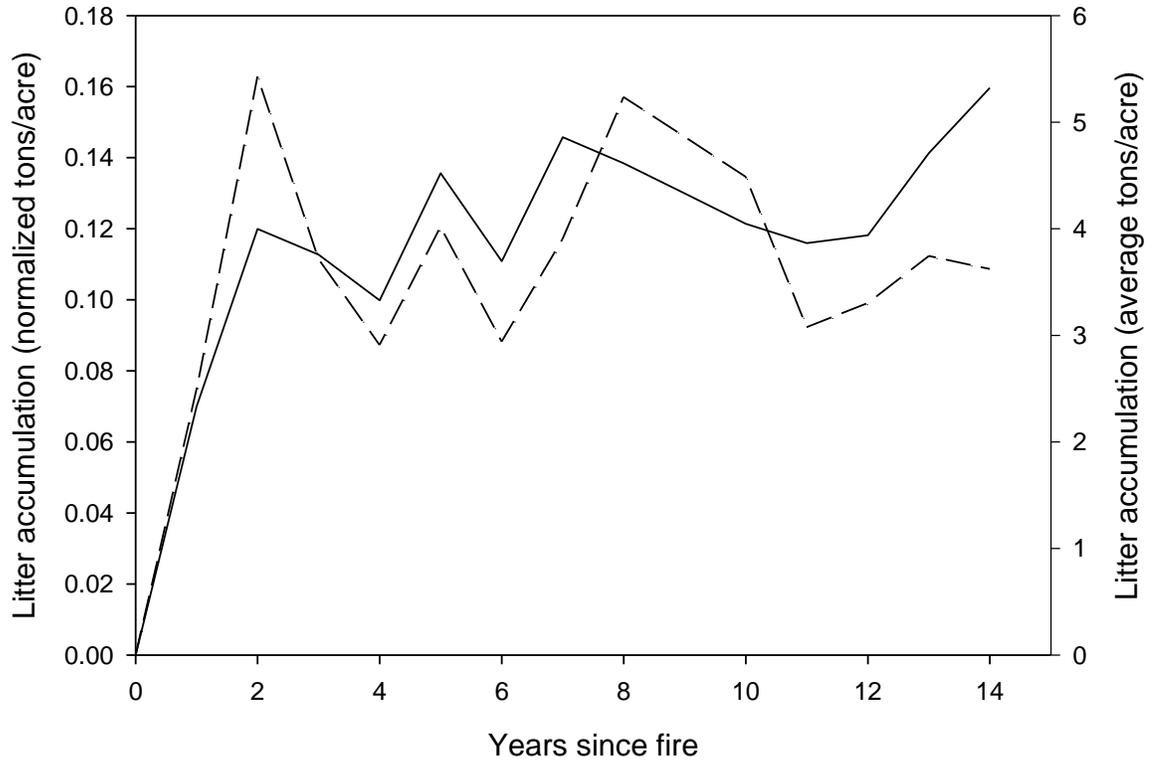


Figure 4.13. Mean plot litter loading measurements (solid line) and normalized litter estimates (dashed line) weighted by stand basal area.

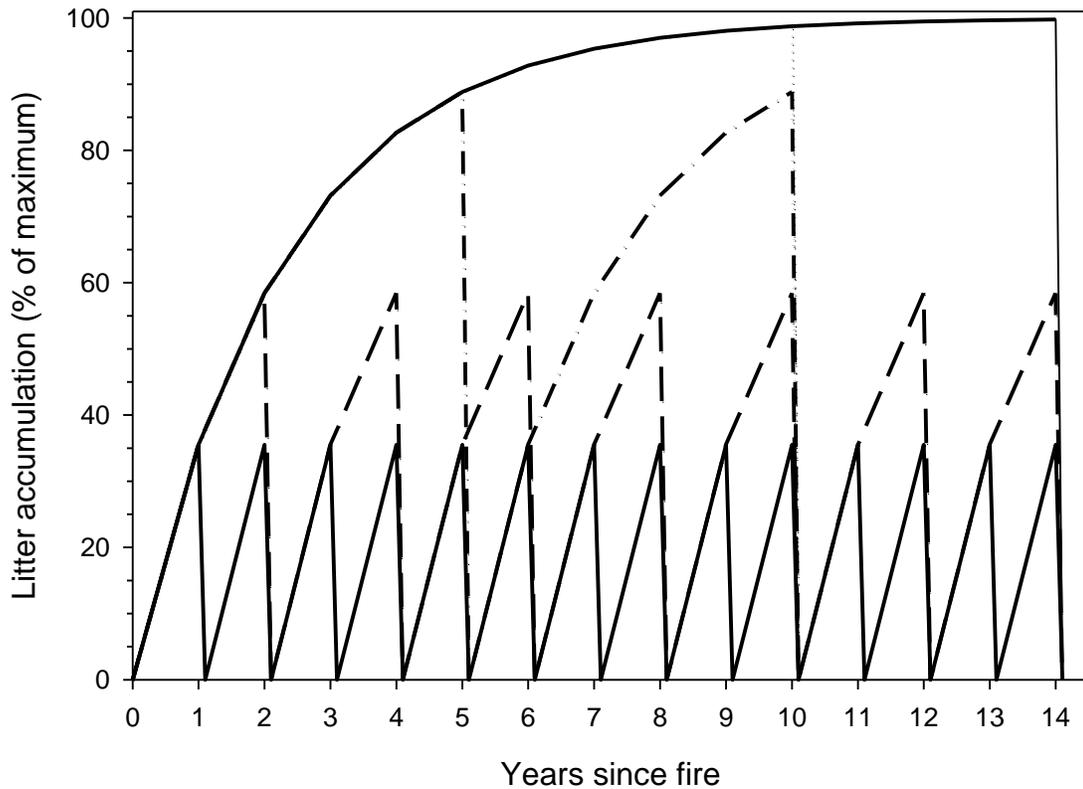


Figure 4.14. Litter accumulation dynamics with removal by fire (or other means) at varying and regular intervals. Annual fire intervals (solid fine line), 2-year fire intervals (dashed line), 5-year fire intervals (dot dashed line), 10-year fire interval (dotted line), and 14-year fire interval (solid bold line) are represented in relation to the percent of litter accumulated.

Palmer Drought Severity Index and averaged litter measurements (Figure 4.15) appeared to display a direct relationship until about 8 years post-fire, when it begins to show an inverse relationship; however, after further investigation, this proved to be a false assumption because all litter was collected in a single year. Average litter mass was adjusted using PDSI values by time-since-fire. The most recent time-since-fire ($t=1$) of calendar year 2007 had the highest contribution throughout the entire chronosequence series (Table 4.5). Inversely, the last year of time-since-fire sampling ($t=14$) of calendar year 1994 had less than a half of a percent contribution to the total found for those 14 years. These weighted PDSI values are should prove to be more representative of the actual effect that drought has on the forest floor accumulation. Plotting the adjusted PDSI index (Figure 4.16) displayed what is represented by the effect of percent of yearly input in the forest litter. Using the new PDSI values its becomes more clear that only the most recent times-since-fire have a significant changing affect, and as time increases the changing affects of yearly dry and wet events become less of a factor.

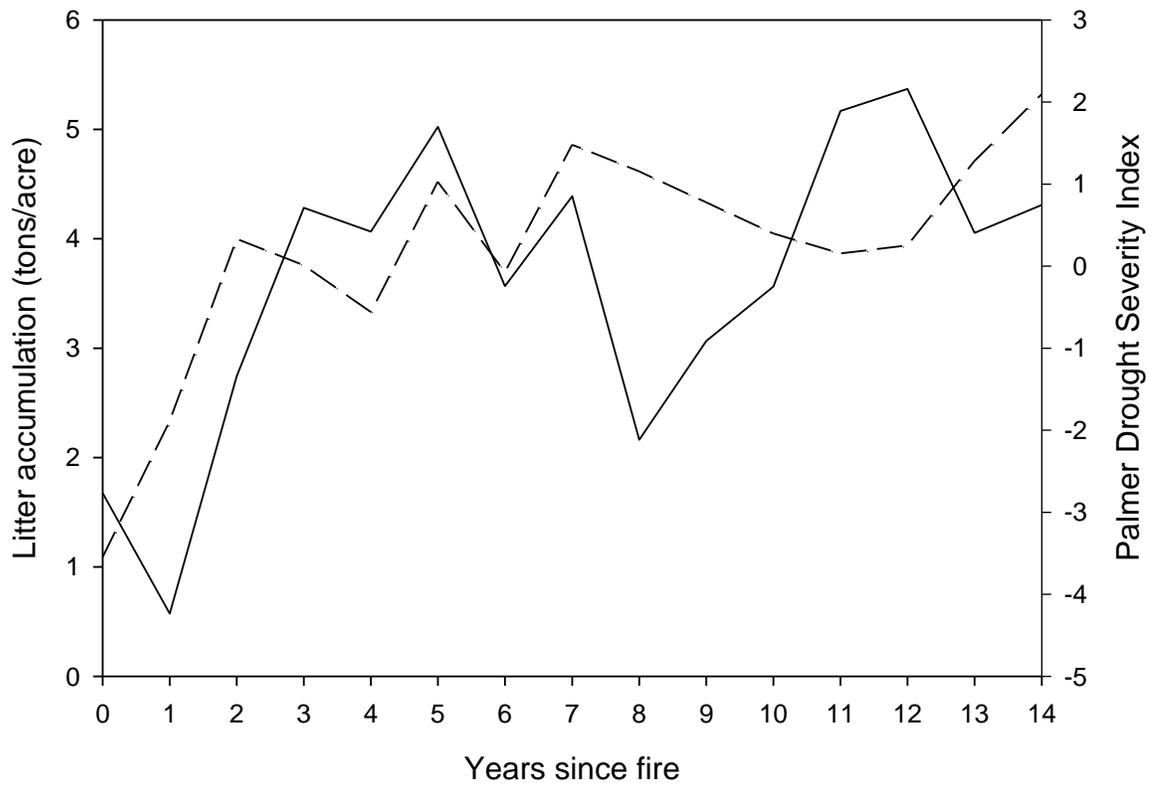


Figure 4.15. Palmer Drought Severity Index (bold solid line) depicting yearly change in drought conditions and averaged yearly litter measurements (dashed line).

Table 4.5. Percent of yearly inputs based on timing of last fire year. The most recent input to the system (i.e. calendar year 2007) holds higher priority and lessens as time-since-fire increases (i.e. calendar year 1994).

	Litter acc. (T/A)	Calendar Year													
		2007	2006	2005	2004	2003	2002	2001	2000	1999	1998	1997	1996	1995	1994
2007	1.889	1.00													
2006	3.107	0.58	0.42												
2005	3.893	0.47	0.34	0.19											
2004	4.400	0.41	0.31	0.16	0.12										
2003	4.727	0.38	0.27	0.15	0.11	0.09									
2002	4.937	0.36	0.27	0.14	0.10	0.08	0.05								
2001	5.073	0.36	0.26	0.14	0.10	0.08	0.04	0.02							
2000	5.161	0.35	0.25	0.14	0.10	0.08	0.04	0.02	0.02						
1999	5.217	0.34	0.25	0.14	0.09	0.08	0.04	0.02	0.02	0.02					
1998	5.254	0.34	0.25	0.13	0.09	0.08	0.04	0.02	0.02	0.02	0.01				
1997	5.277	0.34	0.24	0.14	0.09	0.08	0.03	0.02	0.02	0.02	0.01	0.01			
1996	5.292	0.34	0.24	0.13	0.10	0.07	0.04	0.02	0.02	0.02	0.01	0.006	0.004		
1995	5.302	0.33	0.25	0.13	0.10	0.07	0.04	0.02	0.02	0.02	0.01	0.004	0.004	0.002	
1994	5.309	0.33	0.25	0.13	0.09	0.08	0.04	0.02	0.02	0.01	0.01	0.01	0.006	0.002	0.002

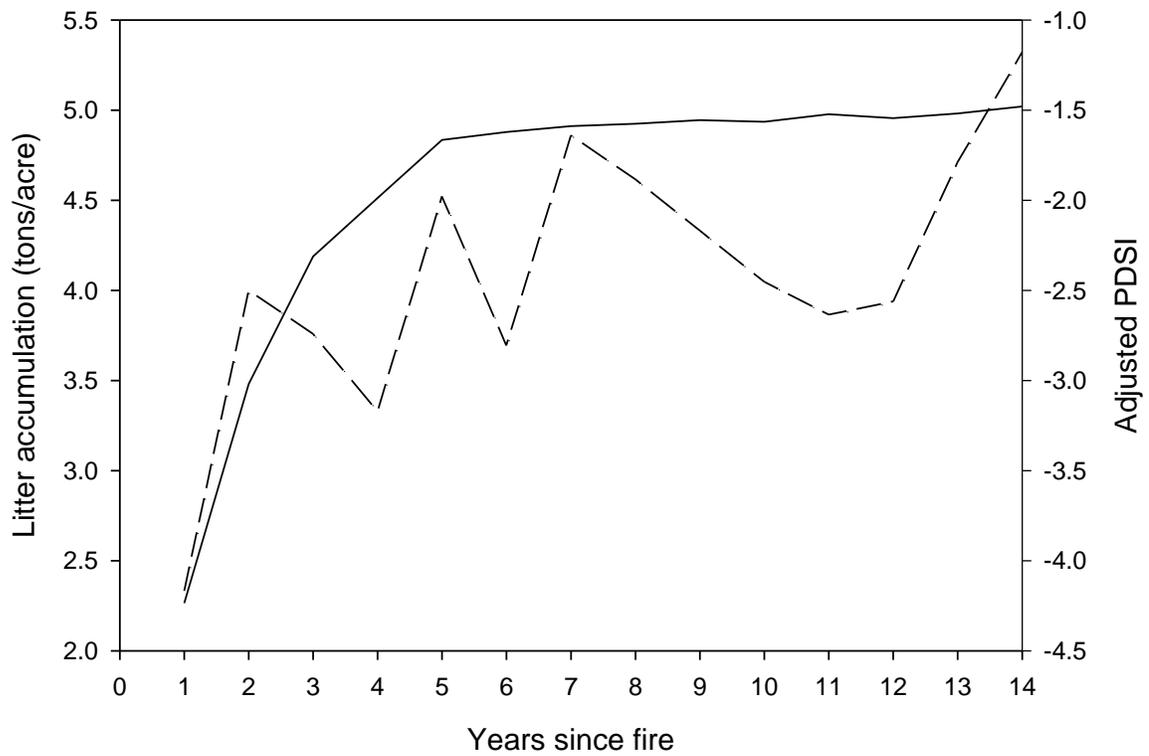


Figure 4.16. Adjusted PDSI values (bold solid line) for Alabama climate division 4. Adjusted PDSI values were based on total percent contribution for each year included through time and begin at one year-since-fire based on extent of PSDI data set. Actual litter measurements per year are represented by the dotted line with the modeled decay rate from Olsen's equation being represented by the dashed line.

CHAPTER 5
DISCUSSION

Fire variability in northeast Alabama

Prescribed fire techniques used in southern montane systems are typically justified based on fire history data (historical documents, charcoal sediments, and modern tree-rings) produced in the Coastal, Piedmont, or upper Appalachian systems. The use of fire history data from a different ecosystem influenced by a different climate (Lafon et al., 2005) can have significant negative effects on re-establishment, management, and site productivity. While paleofire information is relatively abundant in the western U.S., this information is underrepresented in the eastern U.S. and even more so in the Southeast. Southeastern fire history information through pre-EuroAmerican settlement fire scar records is not easily obtainable but is urgently needed to guide restoration of longleaf ecosystems.

Longleaf pine is perhaps one of the best recorders for fire in the southeastern U.S. due to its thick bark, high resin content, and its wide range of historical distribution. It has superior resistance to decay and has the ability to record a high number of fire events. The potential to live for many centuries also increases the ability to detect regional climate signals. These attributes of longleaf make it possible to determine the season of individual fires and fire return intervals.

The initial scarring of longleaf pine boles requires considerable heat to penetrate the thick bark. The extreme temperature variability within a fire suggests that initial scarring of trees can be highly variable (Henderson, 2006). When a single fire event is

recorded on one tree within the fire compartment, this is typically a good indicator of a fire that had both low severity and intensity, which was common on shorter intervals throughout the mid-1700s to late 1800s.

There were few fires recorded prior to 1650; however, there were also fewer remnants available from this time period at both Choccolocco and Brymer Mountain sites. Based on the location of these older remnants near ridgetops, it appears likely that fires during this early period were attributable to natural ignitions (i.e., lightning). At the Choccolocco Mountain site, only two remnant samples had fire scars (n=6) dated to the post-fire suppression era, and these were due to fire management practices of the past 20 years. In addition, no remnants were found at the Brymer Mountain site that covered this same period. This is possibly an indicator of the effects of the fire suppression era on the forest.

In contrast to the earliest and latest portions of the fire chronologies, near annual burning was present at both Choccolocco and Brymer Mountain sites from the early 1700s to the early 1900s. The majority of fire events were of low severity (low percentage of trees scarred throughout the site) and low intensity (minimal cambial tissue damage). This indicated that fires moved quickly through the sites and had low concentrations of fuels in the >1 hr classes that would lead to smoldering and increased cambial damage.

The similarities in fire return intervals between the two sites can be attributed to similarities in slope position and aspect (Pyne et al., 1996). These slopes are predominately south- and west-facing. In comparison to slopes that are predominately

north-facing, south and southwest slopes have longer durations of solar exposure and decreased canopy cover, which reduce the amount of fuel moisture (Clark, 2003) making the areas more prone to carry fire. It is interesting that, despite the distance between the sites, they exhibited relatively similar fire regime patterns. This can be explained simply by their locations—both sites lie in close proximity to well-established travel corridors. This might also explain the modest change in fire frequency upon EuroAmerican settlement in 1832 in northern Alabama (Griffith, 1972; Shankman and Willis, 1995), and reduction of fire events upon the Talladega National Forest establishment in 1936 (About us: Forest History). In addition to their proximity to well-traveled pathways, it is highly likely that these two sites were historically located within the same fire compartment. A fire compartment is defined by Frost (2000) as “an element of the landscape with continuous fuel and no natural firebreaks, such that ignition in one area of the element would be likely to burn the whole”.

Fire seasonality

The seasonality of fire is related to ignition sources as well as fuel characteristics and loading (Henderson, 2006). The majority of all fires at the study sites occurred after latewood formation, indicating that the trees were dormant at the time of these fires. Although late spring and summer fires did occur at both sites, the percentage of trees scarred were low for these fire events due to fuels typically being green and suppressing preheating.

The primary ignition source of late spring and summer fires was probably lightning strikes. Data available since the 1950s indicates that June and July are the peak months for thunderstorms and lightning strikes (Schaefer and Edwards, 1999) in the Southeast. In contrast, the majority of dormant season fires can be attributed largely to anthropogenic ignitions, which were probably the most important source for maintaining recurring fires. Anthropogenic ignitions play an integral role in fire history (Stambaugh and Guyette, 2006). Many anthropogenic ignitions can be attributed to times of war between Native American tribes, French, English, and Spanish settlers (Griffith, 1972), the clearing of land for cultivation and pasture (Early, 2004), and herding and hunting practices (Pyne, 1982). Historical accounts indicate Native American and Euro-American settlers ignited fires in the winter and early spring (Henderson, 2006), which largely shaped the forest structure and stand composition pre-1900s; however, the cessation of these practices and the advent of active fire suppression after the 1930s has altered the fire regime and, consequently, forest composition.

Changing fire regimes

Natural ignitions accounted for very few fire events in the fire chronologies, as most fires occurred during the dormant season. Anthropogenic ignitions were the more likely source for the majority of fires at the study sites, and have historically dominated northeastern Alabama and, to a larger extent, the southeastern United States. Historical human population and industry are defining factors when considering the reasons for variation in fire regimes (Pyne, 1982). For example, when populations are low, mean fire

return intervals are typically longer, and when population increases fire return intervals generally are shortened (Stambaugh and Guyette, 2008).

The naval store industry had a significant impact on the fire regime and seasonality within Talladega National Forest. Fire was a particular danger around turpentine trees, and could potentially wipe out a farmer's entire crop of trees. Crop trees were meticulously cared for and litter was removed from tree bases (Early, 2004). This practice could reduce the potential for injury from fire, and increase the possibility that fire events were not recorded by the tree. This effect could have influenced the recording of fire for up to several decades, depending on the length of time the area was used as a crop for naval stores. There is little doubt that the turpentine industry had widespread negative effects on the montane longleaf forests; however, the greatest negative impacts on these forests would come at the hands of loggers. The montane longleaf system was treated much like any other forest of the late 1800s. Cut-and-run loggers marched through the territory leaving undesirable material behind, along with increased woody debris and fuels (Early, 2004).

Fire regimes of the 20th century appear to support fire intervals that are longer and allow more fuels to accumulate. In recent studies where fire has been reintroduced, tree mortality has been correlated with canopy foliage and branch damage (Varner et al., 2005). The increased fuel loading in these long-unburned areas is a plausible cause of canopy damage, root damage, and excessive mortality following reintroduction of fire. Deep organic horizons accumulate around the bases of longleaf when long periods of fire free intervals occur, allowing roots to begin moving upward to the surface; here the fine

roots can take better advantage of the vast amounts N, P, and K that are being leached from the surface into the soil. The reintroduction of fire will kill these roots that lie close to or within the duff layer. This will hinder growth and nutrient uptake of the tree, and lead to a higher mortality rate than existed prior to the 1900s.

The most influential effects on longleaf pine regeneration and sustainability do to fire regime existed prior to EuroAmerican settlement of the mid 1800s. During this period, fires were frequent and typically of low severity which maintained the longleaf pine component. Due to the short fire return interval, fuels would not have had the length of time to accumulate to levels where intensity and severity would affect tree mortality. The pre-EuroAmerican MFI of 3 years is relatively similar to the EuroAmerican settlement MFI of 2.5 years; however, burning intervals nearing 2 years would have negative effects on regeneration due to the longleaf sapling stage being highly vulnerable to damage, provide higher probability of soil erosion, and loss of nutrients from decaying fuels.

The short fire return intervals found by this study for both pre- and post-EuroAmerican settlement are very similar to those found throughout the coastal longleaf pine ecosystem; however, differences exist in the seasonal distribution of fire events. Throughout this study, the majority of fires occurred within the dormant season, which indicates a higher probability of human ignitions. In the coastal region, fires can occur throughout the entire year, but occur most often in the growing season (Henderson, 2006). Growing season fire scars are most often caused by lightning ignitions. The

frequency of thunderstorm events and cloud-to-ground lightning is higher within the southeastern coastal plains and decreases as one moves inland.

The addition of more fire history sites within the montane longleaf region would strengthen these initial findings and provide a better understanding of temporal and spatial fire regime dynamics specific to this ecosystem. More information on the historic roles of humans, topography, and climate in this fire-dominated landscape would provide greater insight into the ignition sources and conditions under which historic fire intervals occurred. Human population density and topography are important variables for understanding the changes associated with anthropogenic fire regimes (Guyette et al., 2006). Climate change and drought have been shown to affect the periodicity and spatial extent of wildfire at short and long temporal scales in the western United States (Brown and Sieg, 1996; Caprio and Swetnam, 1995; Donnegan et al., 2001) but few such studies exist for the eastern United States.

Forest litter accumulation

The role of litter accumulation and decomposition has been studied extensively in temperate forests and grasslands; however, few such studies have been conducted in longleaf ecosystems (Hendricks et al., 2002). The methodology employed in this study for determining accumulation rates is based on a theoretical approach to define general patterns associated with fire and fuel accumulation. Values obtained using the approaches outlined in the previous chapters are only estimates of possibilities that can occur within the montane longleaf ecosystem.

The best model estimate concerning litter decay and accumulation in the Talladega Upland region was based on Olsen's equation of litter loading measurements and averaged litter mass for maximum accumulation of year 4 through 14 of time-since-fire. This estimate was a better representative due to its incorporation of actual forest fuels data and averaged predictor of maximum accumulation. Other decay rates, based on year 14 time-since-fire maximum accumulation, over-predict the accumulation of litter shortly after 3 years post-fire. This is likely the result of yearly variation in stand basal area, due in part to the management efforts of recent years-since-fire (first year input). Stand basal area increases as less management is applied, increasing to >14 years post-fire (maximum accumulation) where little to no management has occurred. When forest floor measurements were plotted against stand basal areas for each individual year, basal areas remained lower until the last burn year sampled. Basal areas ranged from a mean of 28 ft² at few years post-fire to 53 ft² at maximum years since fire. The increase in basal area has a larger effect on the models using estimated decay constants due to increases in litter deposited on the forest floor for maximum mass accumulation versus the lower deposits from low stand basal areas of more recent time-since-fire years.

Litter accumulation estimates from the logistic regression exponential growth-to-maximum (EGTM) model provided an accumulation curve that reflected all litter measurements. The EGTM model depicts litter accumulating and reaching a steady state much faster than the other models which use decay rates. Use of this model allows for higher degrees of freedom and larger sample sizes. This model estimates 99 percent litter

accumulation 4 years post-fire, four years difference from the climate driven model and averaged maximum accumulation model.

This study's EGTM model shows that more than 90 percent of the total is achieved within 3 years. When compared with biannual burning intervals, we could expect that no more than 88 percent or 3.2 tons/acre would be achieved between fires. Use of this information compared to previous models would calculate that any burning intervals longer than 3 years would not impose a large change in the amount of litter mass on the forest floor; however, factors such as decay and duff accumulation are not represented in this model but do have significant impacts on fire behavior.

Increased strength (i.e. increased number of observations) of time since fire and longer fire records could provide better understanding of litter inputs and help to reduce the variance, adding flexibility to the model's curve. Collection information pertaining to timing of last burn (e.g. season and month) or use of permanent long-term plot data would reduce the uncertainty of between year input and provide "true" measurements for litter consumption and amount remaining directly post-fire (year 0).

The main difference between the litter accumulations models presented in chapter 4 is between 0 years and 1 year since fire. The modeled litter accumulation from Olsen's (1963) decay constant assumes total combustion and consumption of measurable litter mass; however, this effect usually only occurs in the most intense fires where soil sterility is possible. The non-linear all plot data model relies on litter remaining after fire passes through, therefore has a beginning value estimated around one ton per acre. There are many reasons to consider both models correct. The differences between the starting years

are in a large part due to fire intensity. Considering, fire intensity can only be measured during a fire; we can set a wide range of outcomes in how much litter is left post fire event.

Factors that may influence the modeling of litter dynamics include the placement of individual plot locations and soil characteristics associated with each plot. The fluctuation of measured observations within each individual year could be attributed to many ecological factors, including recent mortality within the stand, tree health decline due to insect infestation, and abundant herbaceous growth. Transects were located in areas where the least amount of disturbance was likely to occur; however, there were many occasions where this could not be achieved. Samples were collected based on the assumption that each plot would have an equal or opposite plot that would offset over- or under-measurements, and that plot would be located along the same transect. For example, a dying or recently dead tree can significantly influence the amount of annual litter accumulation observed within one plot; however, when averaged with data from the entire transect and then averaged based on year-since-fire, the higher measured accumulation of that individual plot has less influence.

Soil nutrient levels in different areas of the forest can affect the stand basal area and composition, and affect the degradation of litter that has accumulated on the forest floor. The depth at which litter was sampled excluded the zone of partially decomposed material; however, there is a high likelihood that minimal amounts of material from this zone were incorporated within the layer collected. When selecting locations for litter

accumulation sampling, it is important to consider sampling depth and how variations in soil properties affect the conditions of the forest floor.

The estimated litter accumulation rate derived from Meentemeyer's decomposition (k) equation of mean annual AET and lignin contents was not included as being a final predictor model. This model tended to overestimate mean litter loadings faster and did not incorporate biological factors that play a large role in the production and residence time of litter within a forest. Another reason that Meentemeyer's decay constant tended to over-predict the actual forest measurements could involve the lignin concentration data, which were derived from studies examining elevated CO₂ and nitrogen fertilization effects and not based in a natural setting. Measurements of lignin concentration were also made on sapling longleaf pines. Age variation could play a major role in the concentration of certain structural and chemical properties of the leaf litter (Kane et al., 2008).

The extreme overestimation and steep sloping curve of Meentemeyer's climate-based model in the first 8 years post-fire is a defining factor leading to its exclusion from the final model. Although Olsen's model of maximum averaged litter accumulation requires more time to reach its asymptote than the EGTm, it more closely represents values of actual litter measurements from the forest floor and also incorporates litter decomposition. This decay rate maintains a lower degree of slope in the initial accumulation phase of where it under-predicts the first 2 years and begins to predict stabilization at year 8.

Varner (2005) initiated a study to compare mortality rates due to fire in long-unburned longleaf pine forests with predictions made by the First-Order Fire Effects Model (FOFEM; Reinhardt, 2003). He found that mortality rates differed from model predictions across three different fuel moisture regimes. Mortality rates were predicted equally by FOFEM across all duff moisture contents, and observed mortality increased as duff moisture content decreased. FOFEM predicted no duff consumption in wet and moist burns, and only 8.1 percent in dry duff; however, in the observed burning study, duff consumption averaged 5 percent in wet burns, 14.5 percent in moist burns, and 46.5 percent in dry moisture content burns. This is an important finding relating to total fuel accumulation and loss.

Hendricks and others (2002) found that longleaf pine needles elevated within a wiregrass coastal system obtained decay rates of 0.052 yr^{-1} . The measurements found for this elevated system are more than 50 percent lower than surrounding litter found closer to ground level. The decay rate of elevated longleaf litter is among the lowest decay rates in published literature. Through the use of the decay rates developed by Hendricks and others (2002), an accumulation model would produce a much different curve than found in this study; however, it is unlikely that more than 30 percent of the total litter found near or in the forest floor would be consistent with this very slow decay constant. In addition to skewing accumulation models, it is very likely that modeling fire behavior in this type of system would pose many challenges due to varying degrees of intensity and rate of flame spread.

In this study's model, litter accumulation rates stabilize after 8 years of litter build-up; however, this model does not predict changes in duff layers that exist below the surface fuels. The continual annual input to the surface fuel and the approach of a chronosequence make determining surface fuel-to-duff ratio difficult. A future study based on long-term measurements of litter depth might provide better understanding of this changing ratio, where smoldering fires can cause catastrophic damage to a tree's root system and eventually lead to mortality.

While wildfires involve a combination of factors such as topography, fuels, and weather (Intermediate 8), only fuels may be altered to affect future fire risk and behaviors. Current fire management plans employ prescribed burning and wildfire to control vegetation, competition, and dangerous fuel loadings (Bowman, 2008). Frequent burning limits the amount of fuels on the forest floor, thereby, decreasing the potential risk for severe and catastrophic fire events.

Although this method of interpreting fuel accumulations is more of a theoretical approach, there is viable information that can be extracted and applied to management schemes. Development of an accumulation curve that would be more representative of actual litter measurements would certainly have more applicability, in addition to the benefits of increased sample sizes and a longer time series. There is no doubt that if this study had been a "long-term" look at fuel build-up and accumulation measured every year for the 14 year duration, the results could be matched with less variability, and climatic factors could be included in the creation of a forecasting model.

Early in this study, climate was thought to have a larger role in the total accumulation when compared to the timing of post fire frequencies. Palmer Drought Severity Index (PDSI) and averaged litter measurements appeared to display a direct relationship until about 8 years post-fire, when it began to show an inverse relationship; however, after further investigation, this proved to be a false assumption because all litter was collected in a single year and contain a multitude of PDSI values associated with total litter.

The use of adjusted PDSI values showed that only the most current years-since-fire could possibly affect litter mass; however, this method of weighting was largely ineffective because it was intercorrelated with the modeled litter accumulation curve and the values remained conflicted in the temporal scale. The most accurate way to depict climate variables within litter accumulation is to collect yearly data on fixed plots over a long-term study. This could potentially remove temporal disagreements between increasing litter mass and climate variables. Using yearly climate data and litter inputs it becomes easier to determine lag based effects. Ecologically, the current year's climate (in this case PSDI) usually affects future year(s) growth patterns, decay rates, and litter fall dynamics.

Periods of longer fire intervals can lead to major concern as forest fuels build up to levels higher than historically existed (Stambaugh et al., 2006), creating the possibility for higher intensity fires than the landscape historically experienced (Early, 2004). The variability that occurs from annual burning to 5-year fire intervals is more than double the accumulation. The interaction of forest litter and fine fuels at varying fire frequencies, in

a fire-dominated landscape, is critical information that may one day help in setting management objectives for restoring and maintaining montane longleaf ecosystems, and reduce mortality rates when reintroducing fire to the landscape.

CHAPTER 6

SUMMARY & CONCLUSIONS

The purposes of this study were to (1) describe the role of fire in montane longleaf pine ecosystem, (2) reconstruct historical fire events from fire-scar data and model the pre-EuroAmerican and EuroAmerican settlement mean fire return interval of the forest, (3) characterize the fine fuel loading patterns and decay rates in relation to fire disturbance events using a chronosequence approach, and (4) determine the most appropriate fire regime to address fuels management and the sustainability and regeneration of longleaf pine.

Historically, fire has been a constant and vital influence throughout the Southeast and remains so today. The longleaf pine sites within the study area require fire for a number of reasons; for without recurring fire events, longleaf pine stands would move into mid and late succession states where hardwoods dominate. These reasons include, but are not limited to, stand regeneration through scarification of soil, removal of woody biomass for faster decomposition and nutrient release, removal of competition, and promotion of healthy growth for deterrence of pathogens and insects. These historic fires were maintained by humans for their personal benefit but also benefited the forests they lived within.

The fire regimes represented in this study were separated based on estimates of EuroAmerican settlement throughout northern Alabama. These estimates vary from 20 to 100 years based on settlement criteria. For example, Griffith (1972) indicates settlement

beginning in northern Alabama around 1732, with the grant of Georgia to Oglethorpe and his associates, Shankman and Willis (1995) identify settlement in 1832 based on General Land Office surveys, and Frost (2006) indicates settlement beginning around 1850 with the constraint of more than 2 people per square mile. Based on historical records of Native American movement through and removal from northern Alabama, the best estimate of settlement was provided by Shankman and Willis (1995) using General Land Office survey data for 1832. Although some estimates of early settlement may not be clearly defined, the fire regimes associated with pre-EuroAmerican and EuroAmerican settlement do not vary to a large degree.

Despite historically short fire return intervals, it is unlikely that a fire regime that was once present within a stand can be used presently, without some form of stand manipulation. The most influential burning regime can only be determined once the stand has been returned to historical composition. It is at this point in the stand dynamics that we may be able to apply the historic pre-EuroAmerican settlement fire regime and reintroduce a steady state where regeneration of longleaf is continually replacing longleaf that die.

Fire regimes that dominated pre-EuroAmerican settlement could prove to be more beneficial to the ecosystem and its processes than the prescriptions that are currently being used. A biannual burning regime suggests roughly 60 percent of the total accumulation will build up on the forest floor. Currently, the accumulation per year-since-fire effect on fire intensity and severity are unknown; however, there is evidence to support that there is a maximum point of litter accumulation and number of years where

mortality of feeder roots will occur due to higher severity fires which can have lag based mortality effects on the tree (Cipollini, 2006; Varner et al., 2005).

This study represented a unique opportunity to examine the disturbance history and fine litter dynamics associated with fire disturbance in montane longleaf ecosystems. Small tracts of remnant old growth longleaf pine still exist throughout the mountainous area where longleaf pine once dominated. Understanding the factors that promote their survival is a key to their success. Although fire effects and litter accumulation dynamics are only a small part of the broader ecological scene, they are vital pieces of information. Currently, historic fire regimes are being used to set targets for prescribed burning and management practices. Once a target stand level is achieved, managers will be able to manage and sustain stands based on litter accumulation. Using litter accumulation data can also provide burning prescriptions for areas being managed for an optimum level of fuel loading and desired fire behavior. Even with relatively general information about litter decay and accumulation, decisions about forest management and prescribed burning activities are better informed.

Though based on regionally specific data from the Talladega Upland region, the litter accumulation and decay estimates presented in this study are generalized and do not take into account interannual variability effects (e.g., climate, fire effects (partial litter consumption), and litter production). The estimates and equations provide a context with which to consider fuels management while incorporating an understanding of historic fire regimes, and may provide a foundation for a more refined understanding of fuel and fire interactions.

LITERATURE CITED

- About us: Forest History. *Online Posting*. USDA Forest Service National Forests in Alabama. Accessed 11 Apr. 2009.
<http://www.fs.fed.us/r8/alabama/aboutus/forest_history.shtml>
- Bailey, R.G. 1995. Description of the Ecoregions of the United States. USDA Forest Service, Miscellaneous Publication 1391, Washington, D.C., 108 pp.
- Bale, A.M. and M.C. Stambaugh. 2008. Eastern OLDLIST. A database of maximum tree ages in eastern North America. *Online database*. Accessed 2 Apr. 2009
<<http://people.eku.edu/pedersonn/oldlisteast/>>.
- Bale, A.M., Guyette, R.P. and M.C. Stambaugh. 2008. Fire regime of a montane longleaf pine ecosystem, Alabama. Pages 54-59 *In*: Third Montane Longleaf Conference Proceedings. March 11-12. Auburn, AL.
- Bonnicksen, T.M. 2000. America's Ancient Forest: From the Ice Age to the Age of Discovery. John Wiley & Sons, NY.
- Boring, L.R., J.J. Hendricks, C.A. Wilson, R.J. Mitchell. 2004. Season of burn and nutrient losses in a longleaf pine ecosystem. *International Journal of Wildland Fire* 13: 443-453.
- Bowman, Chris. 2008 "Forest Service explains its 'let it burn' policy." *The Sacramento Bee* 18 Jul. 1B.
- Boyer, W.D. 1983. Variations in height-over-age curves for young longleaf pine plantations. *Forest Science* 29: 15-27.
- Boyer, W.D. 1990. *Pinus palustris* Mill. Longleaf pine. Pages 405-412 *In* Burns, R.M. and B.H. Honkala, tech. coords. *Silvics of North America, Vol. I, Conifers*. USDA Forest Service Agriculture Handbook 654, Washington, D.C.
- Brockway, D.G. and C.E. Lewis. 1997. Long-term effects of dormant-season prescribed fire on plant community diversity, structure and productivity in a longleaf pine wiregrass system. *Forest Ecology and Management* 96: 167-183.
- Brown, P., C.H. Seig. 1996. Fire history in interior ponderosa pine communities of the Black Hills, South Dakota, USA. *Int. J. Wildland Fire* 6 (3), 97-105.
- Carroll, W.C., P.R. Kapeluck, R.A. Harper, and D.H. Van Lear. 2002. Background paper: historical overview of the southern forest landscape and associated resources. Pages 583-

606 *In* Wear, D.N. and J.G. Greis (Eds.), Southern Forest Resource Assessment. USDA Forest Service GTR SRS-53.

Caprio, A.C., T.W. Swetnam. 1995. Historic fire regimes along an elevational gradient on the west slope of the Sierra Nevada, California. In: Brown, J., R. Mutch, C. Spoon, R. Wakimoto. (Tech Coord.), Proceedings: Symposium on Fire in Wilderness and Park Management: Past Lessons and Future, Missoula M.T.

Chapman, H.H., 1922. A new hybrid pine. *Journal of Forestry* 20:729-734.

Cipollini, M. 2006. The Berry College Longleaf Pine Project: Progress over the first three years. Proceedings of the 2nd Mountain Longleaf Conference, Mount Berry, GA. November 18-19. Longleaf Alliance Report 9:9-12.

Clark, Stacy L. 2003. Stand Dynamics of an old-growth oak forest in the Cross Timbers of Oklahoma. Ph.D. dissertation. Oklahoma State University.

Cook, E.R., D.M. Meko, D.W. Stahle, and M.K. Cleaveland. 1999. Drought reconstructions for the continental United States. *Journal of Climate* 12:1145-1162.

Cook, E.R. and R.L. Holmes. 1984. Program ARSTAN and Users Manual. Laboratory of Tree-Ring Research, University of Arizona, Tucson.

Cook, E.R. 1985. A time series approach to tree-ring standardization. Ph.D. dissertation. University of Arizona, Tucson.

Cook, E.R. 2002. Turbo ARSTAN software: version 3.6. Tree-ring laboratory, Lamont-Doherty Earth Observatory, Palisades, NY.

Craul, P.J. 1965. Longleaf pine site zones. Final Report FS-SO-1103. USDA Forest Service, Southern Research Station, Auburn, Alabama.

Croker, T.C., Jr. and W.D. Boyer. 1975. Regenerating longleaf pine naturally. USDA Forest Service, Research Paper SO-105. Southern Forest Experiment Station, New Orleans, LA.

Dieterich, J.H. 1980. The composite fire interval—a tool for more accurate interpretation of fire history. *In*: Stokes, M.A., Dieterich, J.H. (Tech. Coord.), Proceedings of the Fire History Workshop. USDA Forest Service General Technical Report RM-81.

Donnegan, J.A., T.T. Veblen, J.S. Sibold. 2001. Climatic and human influences on fire history in Pike National Forest, Central Colorado. *Can. J. For. Res.* 31, 1526-1539.

Earley, L.S. 2004. Looking For Longleaf: the Fall and Rise of an American Forest. The University of North Carolina Press. Chapel Hill, NC.

Entry, J.A., G.B. Runion, S.A. Prior, R.J. Mitchell, and H.H. Rogers. 1998. Influence of CO₂ enrichment and nitrogen fertilization on tissue chemistry and carbon allocation in longleaf pine seedlings. *Plant and Soil* 200: 3-11.

ESRI (Environmental Systems Research Institute). ArcGIS software v 9.1. ESRI, Redlands, CA.

Fritts, H.C. 1976. *Tree Rings and Climate*. Academic Press: London.

Frost, C.C. 1993. Four centuries of changing landscape patterns in the longleaf pine ecosystem. Pages 17-43 *In* Hermann, S.M. (Ed.), *The Longleaf Pine Ecosystem: Ecology, Restoration and Management*. Proceedings of the Tall Timbers Fire Ecology Conference No. 18.

Frost, C.C. 2000. Studies in landscape fire ecology and presettlement vegetation of the Southeastern United States. Ph.D. dissertation. University of North Carolina, Chapel Hill.

Frost, C.C. 2006. History and future of the longleaf pine ecosystem. Pages 9–42 *In* Jose, S., Jokela, E.J. and D. L. Miller (Eds.), *The Longleaf Pine Ecosystem: Ecology, Silviculture, and Restoration*. Springer, NY.

Griffith, L.B. 1972. *Alabama: A Documentary History to 1900*. Tuscaloosa: The University of Alabama Press.

Griffith, G.E., J.M. Omernik, J.A. Comstock, S. Lawrence, G. Martin, A. Goddard, V.J. Hulcher, and T. Foster. 2001. Ecoregions of Alabama and Georgia, (color poster with map, descriptive text, summary tables, and photographs): Reston, Virginia, U.S. Geological Survey (map scale 1:1,700,000).

Grissino-Mayer, H.D. 2001. FHX2- software for analyzing temporal and spatial patterns in fire regimes from tree-rings. *Tree-Ring Research* 57(1): 115-124.

Guyette, R.P., M.A. Spetich and M.C. Stambaugh. 2006. Historic fire regime dynamics and forcing factors in the Boston Mountains, Arkansas, USA. *Forest Ecology and Management* 234: 293-304.

Hardin, J.W. , D.J. Leopold , and F.M. White. 2001. Harlow and Harrar's textbook of dendrology, 9th ed. McGraw-Hill, Boston.

Harlin, W.V., H.T. Wingate, W.S. Hall, H.O. White, J.A. Cotton, W.B. Parker, and R.B. McNutt. 1961. Soil Survey of Calhoun County. USDA Soil Conservation Service, Washington, D.C.

Haywood, J.D., R.H. Stagg, and A.E. Tiarks. 2004. Relationship between Palmer's Drought Severity Index and the Moisture Index of Woody Debris in the Southern Coastal Plain. USDA Forest Service GTR SRS-71. Southern Research Station, Asheville, NC.

Henderson, J.P. 2006. Dendroclimatological analysis and fire history of longleaf pine (*Pinus palustris* Mill.) in the Atlantic and Gulf Coastal Plain. Ph.D. dissertation. University of Tennessee, Knoxville.

Hendricks, J. J., C.A. Wilson, L.R. Boring. 2002. Foliar litter position and decomposition in a fire-maintained longleaf pine-wiregrass ecosystem. *Canadian Journal of Forest Research*, Vol. 32(6): 928-941.

Holmes, R.L. 1983. Computer-assisted quality control in tree-ring dating and measurement. *Tree-Ring Bulletin* 43: 69-78.

Holmes, R.L., R.K. Adams and H.C. Fritts. 1986. Tree-ring chronologies of Western North America: California, eastern Oregon and northern Great Basin with procedures used in the chronology development work including users manuals for computer programs COFECHA and ARSTAN. Chronology Series VI. Laboratory of Tree-Ring Research, University of Arizona, Tucson.

Intermediate Wildland Fire Behavior. 1994. National Interagency Fire Center. Boise, Idaho.

Jenkins, M. 2005. Wildland fire management and planning. *Home page*. Accessed 2 Mar. 2009. Free Online Course Materials. USU Open Course Ware.
<http://ocw.usu.edu/Forest__Range__and__Wildlife_Sciences/Wildland_Fire_Management_and_Planning/unit3.html>

Kane, J.M., J.M. Varner, J.K. Hiers. 2008. The burning characteristics of southeastern oaks: Discriminating fire facilitators from fire impiders. *Forest Ecology and Management*. Vol. 256: 2039-2045.

Kimmins, J.P. 2004. *Forest Ecology: A Foundation for Sustainable Management*. 3rd Ed. Prentice Hall.

Landers, J.L., D.H. Van Lear, and W.D. Boyer. 1995. The longleaf forests of the Southeast: requiem or renaissance?. *Journal of Forestry* 93 (11): 39-44.

Lafon, C.W., J.A. Hoss, and H.D. Grissino-Mayer. 2005. The contemporary fire regime of the central Appalachian Mountains and its relation to climate. *Physical Geography* 26: 126-146.

- Lafon, C.W. and H.D. Grissino-Mayer. 2007. Spatial patterns of fire occurrence in the central Appalachian Mountains and implications for wildland fire management. *Physical Geography* 28: 1–20.
- Lu, J., G. Sun, S.G. McNulty and D.M. Amatya. 2003. Modeling actual evapotranspiration from forested watersheds across the Southeastern United States. *Journal of the American Water Resources Association* 39(4): 887-896.
- Maceina, E.C., J.S. Kush, & R.S. Meldahl. 2000 Vegetational survey of a montane longleaf pine community at Fort McClellan, Alabama. *Castanea* 65: 147–154.
- McCay, D.H. 2000. The effects of chronic human activities on invasion of longleaf pine forests by sand pine. *Ecosystems* 3: 283–292.
- Meentemeyer, V. 1978. Macroclimate and lignin control of litter decomposition rates. *Ecology* 59(3): 465-472.
- Mohr, C. 1896. The timber pines of the southern United States. USDA Division of Forestry, Bulletin No. 13. Washington, D.C.
- National Interagency Fire Center (NIFC). 1994. Intermediate Wildland Fire Behavior. Boise, ID.
- NCDC. 1994. Time bias corrected divisional temperature-precipitation-drought index. Documentation for dataset TD-9640. Available from DBMB, NCDC, NOAA, Federal Building, 37 Battery Park Ave. Asheville, NC 28801-2733.
- Orvis, K.H. and H.D. Grissino-Mayer. 2002. Standardizing the reporting of abrasive papers used to surface tree-ring samples. *Tree-Ring Research* 58: 47-50.
- Olson, J. S. 1963. Energy storage and the balance of producers and decomposers in ecological systems. *Ecology* 44: 322–331.
- Outcalt, K.W. and R.M. Sheffield. 1996. The longleaf pine forest: trends and current conditions. Resource Bulletin SRS-9. Asheville, NC: USDA Forest Service, Southern Research Station.
- Outcalt, K.W. 2000. Occurrence of fire in longleaf pine stands in the Southeastern United States. Pages 178-182 *In* Moser, W.K. and C.E. Moser (Eds.), *Fire and forest ecology: innovative silviculture and vegetation management*. Tall Timbers Fire Ecology Conference Proceedings, No. 21. Tallahassee, FL.
- Pessin, L.J. 1934. Annual ring formation in *Pinus palustris* seedlings. *American Journal of Botany* 21: 599–603.

- Platt, W.J., G.W. Evans, and S.L. Rathbun. 1988. The population dynamics of a longlived conifer. *American Naturalist* 131: 491–525.
- Platt, W .J., and S. L. Rathburn. 1993. Dynamics of an old-growth longleaf pine population. Tall Timbers Fire Ecology Conference. Tallahassee, Florida.
- Predmore, S. A., J. McDaniel, and J.S. Kush. 2007. Presettlement forests and fire in southern Alabama. *Canadian Journal of Forest Research* 37(9): 1723-1736.
- Pyne, S.J. 1982. *Fire in America: a cultural history of wildland and rural fire*. Princeton Univ. Press, New Jersey.
- Pyne, S.J., P.L. Andrews, and R.D. Laven. 1996. *Introduction to Wildland Fire*, second edition. John Wiley & Sons, Inc., New York.
- Reed, F.W. 1905. A working plan for forest lands in central Alabama. USDA Forest Service, Bull. 68, Washington, D.C.
- Reinhardt , E.D. 2003. Using FOFEM 5.0 to estimate tree mortality, fuel consumption, smoke production, and soil heating from wildland fire. Sec. PS-2 in *Proceedings of Fifth Fire and Forest Meteorology Conference*, Orlando, FL.
- Runion, G.B., J.A. Entry, S.A. Prior, R.J. Mitchell, and H.H. Rogers. 1999. Tissue chemistry and carbon allocation in seedlings of *Pinus palustris* subjected to elevated atmospheric CO₂ and water stress. *Tree Physiology* 19: 329-335.
- SAS/STAT. 2002. *SAS Users Guide* 5th ed. SAS Institute, Cary, NC.
- Schaefer, J. T. and R. Edwards. 1999. The SPC tornado/severe thunderstorm database. The 11th Conference on Applied Climatology, Dallas, Texas.
- Schopmeyer, C.S. 1974. *Seeds of woody plants in the United States*. U.S. Department of Agriculture, Agriculture Handbook 450. Washington, D.C.
- Schwarz, G.F. 1907. *The longleaf pine in virgin forest: a silvical study*. Wiley, New York.
- Shankman, D. and K.M. Willis. 1995. Pre-European settlement forest communities of the Talladega Mountains, Alabama. *Southeastern Geographer* 35: 118-131.
- Smith, K.T. and E.K. Sutherland. 2001. Terminology and biology of fire scars in selected central hardwoods. *Tree-Ring Research* 57: 141–147.

Southeast Regional Climate Center. The University of North Carolina-Chapel Hill Geography Department. 20 Feb. 2009 <<http://www.sercc.com/>>

Spivey, L.D. Jr. 1974. Soil Survey of Cleburne County. USDA Soil Conservation Service, Washington, D.C.

Stambaugh, M.C. 2001. Forest canopy gap dynamics in shortleaf pine forests of the Ozark Highlands. M.S. Thesis. University of Missouri-Columbia.

Stambaugh, M.C. and R.P. Guyette. 2004. Long-term growth and climate response of shortleaf pine. Pages 448-458 *in* (add editors) 14th Annual Central Hardwoods Conference Proceedings, Wooster Ohio, March 2004. USDA Forest Service, GTR-NE-316.

Stambaugh, M.C. and R.P. Guyette. 2006. Fire regime of an Ozark Wilderness Area, Arkansas. *American Midland Naturalist* 156: 237-251.

Stambaugh, M.C., R.P. Guyette, K. Grabner, and J. Kolaks. 2006. Understanding Ozark forest litter variability through a synthesis of accumulation rates and fire events. Pages 321-332 *In* Butler, B.W and Andrews, P.L. (Comps.), *Fuels Management- How To Measure Success: Conference Proceedings*. 2006, 28-30 March; Portland, OR. USDA Forest Service GTR RMRS-P-41.

Stambaugh, M.C. and R.P. Guyette. 2008. Predicting spatio-temporal variability in fire return intervals using a topographic roughness index. *Forest Ecology and Management* 254: 463-473.

Stokes, M.A. and T.L. Smiley. 1968. *Introduction to Tree-Ring Dating*. University of Chicago Press, Chicago, IL.

Stanturf, A.J. 2008. *Online posting*. Version 5. 28 Feb. 2008. Southern Research Station, USDA Forest Service. <<http://forestryencyclopedia.jot.com/WikiHome/Wildfire>>

Tangren, C.D. 1976. The trouble with fire intensity. *Fire Technology* 12: 261-265.

Van Lear D.H., W.D. Carroll, P.R. Kapeluck, and R. Johnson. 2005. History and restoration of the longleaf pine-grassland ecosystem: Implications for species at risk. *Forest Ecology and Management* 211(1-2): 150-165.

Van Wagtenonk, J.W. 2007. The history and evolution of wildland fire use. *Fire Ecology Special Issue: Wildland Fire Use* 3: 3-17.

Varner, M.J., J. S. Kush, and R. S. Meldahl. 2003a. Structure of old-growth longleaf pine (*Pinus palustris* Mill.) forests in the mountains of Alabama. *Castanea* 68: 211-221.

- Varner, M.J., J. S. Kush, and R. S. Meldahl. 2003b. Vegetation of frequently burned old-growth longleaf pine (*Pinus palustris* Mill.) savannas on Choccolocco Mountain, Alabama, USA. *Natural Areas Journal* 23: 43-52.
- Varner, J.M. and J.S. Kush. 2004. Old-growth longleaf pine forests and savannas of the southeastern USA: Status and trends. *Natural Areas Journal* 24:141-149.
- Varner, Morgan. 2005. Smoldering fire in long-unburned longleaf pine forests: linking fuels with fire effects. Ph.D. Dissertation. The University of Florida. 111 pp.
- Varner, Morgan J. III, D.R. Gordon, F.E. Putz, J.K. Hiers. 2005. Restoring fire to long-unburned *Pinus palustris* ecosystems: novel fire effects and consequences for long-unburned ecosystems. *Restoration Ecology*, Vol. 13(3): 536-544.
- Vörösmarty, C.J., C.A. Federer and A.L. Schloss. 1998. Potential evaporation functions compared on US watersheds: Possible implications for global-scale water balance and terrestrial ecosystem modeling. *Journal of Hydrology* 207: 147-169.
- Wahlenberg, W.G. 1946. Longleaf Pine: Its Use, Ecology, Regeneration, Protection, Growth, and Management. Charles Lathrop Pack Forestry Foundation, Washington, D.C.
- Walker, L.C. and B.P. Oswald. 2000. The southern forest: geography, ecology and silviculture. Boca Raton, FL: CRC Press LLC.
- White, P.S., S.P. Wilds, and G.A. Thunhorst. 1998. Pages 255-314 *In* M.J. Mac, P.A. Opler, C.E. Puckett Haecker, and P.D. Doran (Eds.), Status and trends of the nation's biological resources. 2 vols. Reston, VA. U. S. Department of the Interior, USGS.