REVEGETATION POTENTIAL OF SLASH PILE BURN SITES IN THE LONGLEAF PINE ECOSYSTEM

by

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(Under the Direction of L. Katherine Kirkman)

ABSTRACT

Restoration of the *Pinus palustris* ecosystem may incorporate removal of encroaching hardwood species, creating wood debris or “slash” that poses a potential wildfire hazard and is piled and burned on-site. We evaluated the response of soils, vegetation, and the soil seed bank to slash pile burns over a chronosequence of time since burn, and established revegetation treatments of *P. palustris* and *Aristida stricta* in recently burned fire scars. We measured lethal soil temperatures during burn and found persistent alteration of soil nutrients at 0-6 years after burn, as well as a drastically reduced seed bank. Recolonizing vegetation in fire scars may not support movement of prescribed fire and thus may allow future hardwood encroachment in these areas. Outcomes of revegetation treatments with native species and topsoil amendments suggest that further restoration of slash pile burn sites to reconnect fire corridors is possible.

INDEX WORDS: slash pile burn, severe fire effects, restoration, longleaf pine ecosystem, *Pinus palustris*, *Aristida stricta*, topsoil amendment

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B.A., Indiana University, 2005

B.S., Indiana University, 2005

A Thesis Submitted to the Graduate Faculty of The University of Georgia in Partial Fulfillment of the Requirements for the Degree

MASTER OF SCIENCE

ATHENS, GEORGIA

2009

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May 2009

ACKNOWLEDGEMENTS

I would like to give my sincere thanks to Dr. Kay Kirkman for her tireless effort, encouragement, and generosity, and for sharing her deep knowledge of the longleaf system. I greatly appreciate the advice and recommendations that Dr. Ron Carroll offered in planning this project, and I am indebted to Dr. Larry Morris for his frequent instruction and enthusiasm. I thank Dr. Joseph O'Brien for giving me the opportunity to quantify extremely intense fires with relative ease.

Ichauway is an amazing and beautiful place and I am grateful to have had the opportunity to live in the woods and work with so many talented people. I want to acknowledge members of the Plant Ecology Lab for their friendship and advice, including Melanie Kaeser, Katie Stuble, Gwen Iacona, Scott Wiggers, Stribling Stuber, Amanda Briant Steen, Christy Cincotta, and Candice Tiu, as well as the many other members who have cycled through and left their mark.

I wish to thank Dr. Mike Conner, Dr. Lindsay Boring, and Kevin Hiers for their generous counsel, as well as Liz Cox, Jean Brock, and Josh Warren for their valuable assistance. I benefited from the shared knowledge of the Conservation staff, including Jimmy Atkinson, Mark Melvin, Scott Smith, Jonathan Stober, and Bobby Bass, and appreciated their willingness to coordinate burn days. Jason McGee deserves acknowledgment for giving his time to help with thermal imaging. I have greatly enjoyed working with David Varnadoe and Susan Bryan, and hope that my thumb is a little greener as a result. I want to thank Sean Sterrett, Carla Atkinson, Lindsey Sargent, and Kathleen Rugel for friendship that spanned the UGA-Ichauway continuum. Rosanne Bohannon, Becky Gay, Cindy Craft, Mary Cobb, Jessica Kelly, Charlotte DeLucca, Larry Ethridge, Jack Mulliford, and David Green also deserve warm thanks. I want to acknowledge Lisa Dean of the UGA Analytical Lab, Patsy Pittman from the Odum School of Ecology for her long-distance help, John Cox and Cody Laird of Lolly Creek Ecological Land Management and Restoration, and Amy Carter and Dana Cheek of NESPAL. I am grateful to Mark Hainds from The Longleaf Alliance for sharing his observations of planting in various site conditions. I would also like to thank Carolyn Kindell and Amy Jenkins of the Florida Natural Areas Inventory, together with Dr. Doria Gordon and Dr. David Sylvia, for supplying me with an extensive response to an inquiry on wiregrass.

I wish to thank my parents, Jeanne and John Creech, and all of my family for their love and encouragement.

This project was supported by the Joseph W. Jones Ecological Research Center, the Robert W. Woodruff Foundation and the University of Georgia.

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PROJECT OVERVIEW

Slash pile burns are used to eliminate unmerchantable wood debris that can result from forestry operations, and which pose future fire hazards. Slash pile burning is frequently implemented in restoration projects following tree thinning or selective removal of undesirable tree species (Korb et al. 2004; Phillips and Waldrop 2008); however, the use of slash pile burning within a restoration context may complicate restoration outcomes due to alterations to vegetation, soils and the soil seed bank (Korb et al. 2004; Haskins and Gehring 2004). In landscape-scale restoration of the fire-dependent *Pinus palustris* Mill. – *Aristida stricta* Michx. (longleaf pine – wiregrass) ecosystem, slash pile burning has been utilized following removal of successional hardwood species that have encroached into the longleaf pine system over previous decades. This study examines the soil alteration and vegetation recolonization evidenced in slash pile burn scars to evaluate the impact they may have on prescribed fire, which is essential to prevent future hardwood succession.

The following sections review pertinent literature on the characteristics of high intensity fires, the impacts on soil physical and chemical traits, soil biota and potential impacts of altered soil conditions on revegetation processes.

LITERATURE REVIEW

Effects of High Intensity Burns on Soils

Fires that consume large piles of woody debris differ considerably from prescribed fires (Certini 2005), such as those used in the management of the frequently burned longleaf pine ecosystem. Typically, prescribed fires in longleaf pine forest are carried quickly across the landscape by the presence of herbaceous fine fuels. Thus, elevated soil temperatures are lower and of shorter duration in controlled burns than in slash pile burns. Heyward (1938) recorded maximum soil temperatures just below the soil surface (0.3-0.64 cm) between 66-79°C for no more than 2-4 minutes during a fire in longleaf pine forest. Consumption of large amounts of fuel such as from slash piles results in much higher temperatures at greater soil depths, largely a consequence of burn duration (Certini 2005). Such burns may last from 2-4 days (J. Stober, personal communication) and produce maximum sustained temperatures between 500-700°C at the soil surface. Instantaneous temperatures can reach 1500°C (Neary et al. 1999). Belowground temperatures during burning can increase for prolonged periods where fuels are heavy. For example, in an Australian *Eucalyptus* slash pile burn, soil temperatures in a dry sandy loam soil were documented at 100°C at 22 cm soil depth for over 24 hours (Roberts 1965).

Fire duration and fire intensity (the rate at which thermal energy is produced by fire) are most responsible for causing changes below the soil surface that impact biological organisms (Neary et al. 1999; Certini 2005). Heat conduction is the mode of thermal energy transfer most likely responsible for increased belowground temperatures during burns of heavy fuels such as slash (DeBano et al. 1998). Short, but intense fires usually cause less impact than fires of longer duration that can conduct heat deeper into the soil. However, the depth that high temperatures are reached is mediated by both soil texture and soil moisture. Generally, dry soil is a poorer medium for heat conduction than moist soil and, consequently, only the top few centimeters of dry soil will typically experience extremely high temperatures during prescribed fires (Certini 2005). In wet soils the presence of water molecules in soil pores can act both to transfer heat deeper than in dry conditions, and can serve as a buffer to high subsoil temperatures until all water molecules are vaporized (occurring at 95°C in soils) (DeBano et al. 1998). Thus, fires of low to moderate intensity may cause less soil alteration in moist rather than dry soils because of moisture's buffering capacity, while high intensity fires are more likely to cause deeper and more severe damage in moist soils (Campbell et al. 1994; Certini 2005).

Physical Traits

Soil texture can also have a significant role in determining depth of heat conduction in the soil. Fine-textured or compacted soils can conduct heat to greater depths than coarser textured soils (Roberts 1965; Seymour and Tecle 2004).

Physical characteristics of soil that may be altered due to prolonged high temperatures include soil bulk density (the dry mass per unit volume of soil), porosity (the volume of pore space in a soil sample per bulk volume of the sample), and hydraulic conductivity (the capacity of soil to transmit water). Heat effects cause the breakdown of soil structure, i.e. the assemblage of soil particles bound into aggregates of certain patterns, for example, blocky or prismatic structures (DeBano et al. 1998; Fisher and Binkley 2000). More aggregated soil structure allows for better movement of water and gases through soil because macropore space is well-maintained. Soil organisms and roots can also move or extend more easily through pores in structurally developed soil (Bardgett 2005).

Another possible physical alteration of soils due to prolonged exposure to heat, particularly in coarse-textured soil, is the formation of hydrophobic layers below the ash layer (Debano et al. 1998). When organic material combusts and steep temperature gradients form in dry mineral soils, organic materials can move deeper into soil and coat or bond with mineral particles. Acting as cementing agents, hydrophobic compounds can reduce hydraulic conductivity and lead to increased water runoff, erosion, and lower plant available water. However, this process may increase soil structure if macropores form that allow water movement to occur and produce more stabilized soil aggregation that reduces erosion and eventually supports plant productivity (DeBano et al. 1998). The most severe development of water repellency occurs at soil temperatures between 176ºC and 204ºC, while hydrophobic substances are broken down and removed at soil temperatures greater than 288ºC (DeBano et al. 1998). In addition to heat, use of heavy equipment on a burn site also may cause soil compaction, particularly on fine-textured or moist soils (Fisher and Binkley 2000).

Chemical traits

Fuel combustion releases cations, particularly Ca, Mg, and K, which can increase soil pH as ash leaches through the soil profile (Certini 2005). Many nutrients become more available at more neutral pH values (Bardgett 2005). When fuel combustion occurs in coniferous forests, the typically acidic soils associated with these ecosystems become less acidic. Soil pH has been found to climb by 1 unit in grassland fires, and by more than 3 units in forest fires (Raison 1979); thus slash piles composed of woody material are expected to raise soil pH. Although ash from these burns contains a large store of cations available for immediate microbial and plant uptake over time, ash is gradually removed from the soil environment by wind or leaching. Leaching can occur more rapidly in sandy soils having larger pore spaces than clayey soils with smaller pore spaces and negatively-charged surfaces which invite cation bonding (DeBano et al. 1998).

The chemical composition of soil is not significantly changed if soil temperatures remain below 100ºC (DiTomaso et al. 2006). Volatilization of nutrients that leads to nutrient loss begins at temperatures ranging from 180ºC for carbon to more than 1000ºC for Ca and Mg (Neary et al. 1999). Nitrogen loss is initiated at 200ºC, and over half of soil N can be volatilized by the time soil reaches 500ºC. Potassium vaporizes at temperatures greater than 760°C, P at temperatures greater than 774ºC, Na above 880ºC, Mg at temperatures over 1107ºC, and Ca beyond 1240ºC (Neary et al. 1999). Volatilization accounts for only a partial loss of nutrient reserves. Nutrients may also undergo chemical changes in which they are converted into more available forms for microbial and plant uptake as a result of fire (Bardgett 2005).

In the case of nitrogen, intense fire and combustion of organic material can increase available forms of N, namely ammonium (NH_4^+) and nitrate (NO_3^-), although total N may decrease due to volatilization losses (Raison 1979). Ammonium forms as combustion reaches temperatures of 100ºC (Raison 1979), while nitrate develops from ammonium later on through nitrification, a process mediated by soil microbes (Certini 2005). Both forms of N are available for plant uptake in soil solution. Yet nitrate, the product of nitrification, is highly soluble and easily leaches out of soil. Ammonium is more apt to bond with available clay minerals and thus resist leaching. Over time, it can be nitrified. For these reasons, sources of available N are typically not long-lasting (Certini 2005). For example, soon after fire, a stand of *Pinus edulis* and *Juniperus spp.* in the southwestern U.S. had elevated soil ammonium up to 50 times greater than prior to fire. After one year, nitrate levels were still much higher than pre-fire conditions but ammonium levels had declines significantly, and after 5 years, increased forms of available N were no longer present, although total N remained increased (Covington et al. 1991).

The possibility of nitrogen loss in ecosystems is a concern because N is typically a limiting nutrient. The ratios of C to N (C:N) in soil will also determine N-mineralization rates by microbes, and thus how much N is available to plants. When C:N in soil is high, microbes release very little ammonium through decomposition, instead using N to grow in population size. When C:N is low, microbes are not N-limited and will begin to release NH_4^+ as wastes that plants can take up (Bardgett 2005). Thus, even available N sources can be withheld from plant use when C sources are comparatively much higher.

Phosphorus is a limiting nutrient in many ecosystems and the fate of P following fire is of considerable interest. Phosphorus becomes orthophosphate with burning, and this form is available for plant uptake while other forms of P are not (Certini 2005). Phosphorus availability is largely dictated by soil pH. When pH is less than 5, P combines with Fe and Al to form Fe and Al phosphates, while at pH greater than 7, phosphate bonds with Ca and is unavailable (Bardgett 2005). Burning of slash piles in coniferous forests that have naturally acidic soils increases soil pH and, thus, can make P more available. Leaching of P is minimal, and P is soon bound with soil minerals (Certini 2005). P-cycling is mediated in ecosystems by microbes, and notably mycorrhizal fungi (Bardgett 2005).

Effects of High Intensity Burns on Biota

<u>Seed Bank</u>

Buried persistent seed banks are often an important component to regeneration in fire-adapted ecosystems. Seeds vary by species in their ability to withstand elevated temperatures

(Baskin and Baskin 1998). For fire-adapted species, fire encourages seed germination, by smoke or heat cues (Wright and Bailey 1982).

The intensity and duration of slash pile fires, however, can create severe temperature gradients through soil that make it unlikely for soil seed banks to survive. Baskin and Baskin (1998) note that the likelihood of seed germination for a variety of species is markedly reduced when temperatures reach anywhere between 50-200ºC and last between 1-5 minutes. In moist soils experiencing high temperature fires, seeds may be steamed as water molecules vaporize, and seeds with higher moisture contents are more likely to be rendered inviable than dry seeds that have hard seed coats (Baskin and Baskin 1998). In dry soils, seeds can generally survive up to 90ºC, and in wet soils up to 70ºC before they are destroyed by heat (DeBano et al. 1998). Since temperatures may reach 100ºC at soil depths of 22 cm under slash piles (Roberts 1965), any seed banks present are likely to be eradicated. Without a seed bank, seed dispersal from the surrounding area will be required for plant re-establishment. Revegetation in severely burned areas may be further hindered by altered soil conditions.

Soil Organisms

Soil organisms can be drastically affected by slash pile burns, because their tissues are unable to sustain the high temperatures. Soil microbes represent an important biological component of soils. Microbes, which include bacteria, fungi, actinomycetes and algae, decompose organic matter using enzymes that convert nutrients into forms available to plants (Bardgett 2005). A variety of other organisms live in the soil, such as macrofauna which move within the litter layer or tunnel through soils, increasing porosity and encouraging soil mixing. Both macro- and microfauna can feed on and regulate other organisms, including microbes.

When soil temperatures increase due to fire, soil organisms can tolerate some change up to a threshold temperature, after which they will be killed. Soil temperatures are typically greatest closest to the surface, where plant roots reside along with many bacteria and fungi that feed on root exudates and decaying organic matter. Plant roots begin to experience mortality between 48-54°C (DeBano et al. 1998), and 60°C is the temperature of instantaneous plant tissue death (Wade and Johansen 1986). Some fungal species can survive in dry soil up to 80°C, and in wet soil to 60°C. AM fungi can persist to 94°C. Likewise, some bacteria can live in dry soil to 120°C, and in wet soil to 100°C. Although organisms living at greater soil depths have a better chance of survival in the presence of fire, elevated soil temperatures associated with burning of slash pile burns often penetrate deep into soils. Thus slash pile burns usually affect more organisms, relative to fires in low or moderate fuels.

Bacteria and Fungi

Fire effects can bring about significant changes to bacterial and fungal soil assemblages, which are among the most important inhabitants of soils for purposes of nutrient availability and cycling. Bacteria and fungi possess some similar functional traits as they both decompose organic material and mineralize nutrients into forms that plants can take up. Fungi are better able to transfer nutrients across distances in soil, via hyphae. Bacteria, in contrast, are relatively immobile or passive-movers in soil, and will stay inactive if resources are lacking in their microenvironment (Bardgett 2005). Following fire, bacteria are expected to recolonize an area before fungi (Bardgett 2005; Jiménez Esquilín et al. 2007), as bacteria are more abundant at the higher pH that often results from fire. Over time, some leaching of basic cations occurs, and revegetation introduces organic inputs including organic acids. As the soil acidifies, fungal populations will increase and even surpass bacterial populations (Bardgett 2005).

An important genus of bacteria in the longleaf system, *Rhizobium,* is a nitrogen-fixer that is commonly associated with the roots of legume species and supplies N to the soil system by converting atmospheric N to available forms for plant uptake (Bardgett 2005). Legumes are especially common and diverse members of vegetation assemblages in the longleaf pine forest, supplying N to soils where it is frequently lost due to volatilization by fire (Hainds et al. 1999). Some arbuscular mycorrhizal (AM) fungi can interact with *Rhizobium* to boost nodulation and N-fixation rates in infected legumes, and also move N from legumes to neighboring plants through hyphae (Bardgett 2005).

Mycorrhizal Fungi

More than 95% of vascular plant genera have mycorrhizal associates (Bardgett 2005). Mycorrhizae can make nutrients such as P and N more available to the associated plant, while receiving C from the plant in return (Bardgett 2005). When soils are P-limited, more plant photosynthetic C has been found to be transferred to mycorrhizal fungi than when soils are not P-limited (Bardgett 2005), suggesting that mycorrhizae are more important when P is lacking. There are three types of mycorrhizae that have different plant associates: ectomycorrhizae associate with many tree species including conifers such as longleaf pine, while arbuscular mycorrhizae are more typically found with herbaceous plants, and ericoid mycorrhizae associate with ericaceous plants (Bardgett 2005).

Arbuscular mycorrhizal (AM) associations have not been well-studied across the southeastern U.S., although limited mycorrhizal potential among some plant species common to longleaf pine forest has been documented. In the dry and nutrient-poor Florida sandhills, Anderson and Menges (1997) evaluated AM associations and the effects of prescribed fire on four herbaceous species, including wiregrass, *Aristida stricta*, and three forb species, *Liatris* *tenuifolia* var. *laevigata*, *Pityopsis graminifolia*, and *Balduina angustifolia.* No mycorrhizal colonization was found in *Aristida*, which was formerly the dominant ground cover species across much of the eastern natural extent of longleaf forest. The shallow and many-branched root structure of *Aristida* differs from the tap root system more typical of a mycorrhizal plant. The forbs generally showed low root colonization (0 - 4 percent) over 10 months for all species except *Balduina angustifolia* (7-38 percent). Other work in the longleaf system has documented AM inoculation of wiregrass roots (Kindell and Alden 1993, unpublished data; Jenkins 2003), the uniform presence of mycorrhizal spores in longleaf pine upland soils (Henkell and Kindell 1991) and the presence of moderate mycorrhizal potential in associated soils (D. Sylvia and D. Gordon, personal communication). Thus the role of arbuscular mycorrhizae in relation to herbaceous species in this system remains unclear.

In the southeastern U.S., slash piles located on old fields and on or near former agricultural food plots may also have elevated soil P due to past fertilizer additions. Such P fertilizer inputs are maintained in soil storage for long periods (Fisher and Binkley 2000; Markewitz et al. 2002), compared to N inputs that are more prone to leaching. Thus, potential benefits that native mycorrhizal additions might provide to plants in promoting their establishment, such as making P more available, may be dependent on land use history. Although fire may remove some P from the soil system, it volatilizes at such a high temperature (774°C) that losses due to burning slash piles are probably minimal (DeBano et al. 1998).

Mycorrhizae are sensitive not only to heat, but to other disturbances as well. Agricultural techniques, such as tillage, disrupt upper levels of soil and mix organic material on which soil organisms feed, altering soil organism assemblages (Bardgett 2005). Continued impacts result when fields lie fallow, causing mycorrhizal abundance to diminish due to the lack of plant structures to support them (Jasper 1994). Arbuscular mycorrhizae have more limited dispersal abilities (belowground only) compared to ectomycorrhizae that are able to release spores aboveground (Allen 1991). Arbuscular mycorrhizae are likely to more slowly recolonize slash pile burn sites compared to ectomycorrhizae.

Vegetation Recolonization

Recolonization Difficulties

The rate of revegetation of slash pile burn sites depends on many variables such as ash deposition, soil alteration including reductions in mycorrhizal and other soil biota, and erosion which removes topsoil. The potential for plant competition may be reduced in burned soils because they are typically not readily colonized soon after burn. Weedy or non-native invasive species may be better able to colonize these fire-disturbed areas than more desirable native species. The presence of ash, though high in nutrients, may impede revegetation, depending on ash depth. Results from a study of slash and burn agriculture indicated that thick ash beds prevented seed germination, whereas thinly distributed ash did not impede germination and actually made nutrients more widely available for plant uptake (Menzies and Gillman 2003). Deep ash layers may result from large slash pile burns. However, the presence of ash may be a temporary condition, occurring immediately following fire and blowing away with wind or leaching through soil within one year or more (DeBano et al. 1998).

Char deposition with burn, however, represents an input expected to remain in soils (unless removed by erosion) for centuries to millennia (Lehmann and Rondon 2006). Char has strong adsorption capacities and a porous structure that fosters microbial populations, including both bacteria and fungi (Zackrisson et al. 1996; DeLuca and Aplet 2008), and may retain some nutrients from leaching, offering long-term nutrient release (Lehmann and Rondon 2006).

Severely burned areas in which vegetation is destroyed experience different temperature regimes over time than that of vegetated areas. Burned surface soils are darker and absorb more solar radiation than vegetated soils (Raison 1979). In addition, diurnal temperature changes are greater in burned soils than in vegetated areas where vegetation can help insolate soil and reduce temperature fluctuations (Fisher and Binkley 2000). Lack of cover causes greater seasonal temperature fluctuations as well, reaching higher temperatures in summer, and colder temperatures in winter. Such differences in soil temperature may impact seed germination and establishment.

Fires in heavy fuels may produce bare spots that remain for years after fire events. In a Missouri oak savanna situated on cherty silt loams, coarse woody debris in the form of fallen tree boles had burned 3-6 years before, creating "log shadows" where no vegetation returned (Rhoades et al. 2004). Although nutrients such as P, K, Ca, Mg, and NO_3^- were higher in log shadows than the surrounding savanna (characteristics which would be expected to encourage revegetation) C:N was high and mineral N was low, suggesting that strong N-limitation was evident.

Following fire, bare soils may lack inoculation sources, inhibiting AM establishment on seedlings. For AM species having accessible inoculation sources, seedling inoculation may occur soon after seed germination. For example, in chalk grassland in England, 11 of 12 native species developed mycorrhizal associations 7-10 days after germination (Fenner and Thompson 2005). Thus, when severe temperatures eradicate inoculant sources, AM plants may experience lower survivorship and seed production (Korb et al. 2004).

Burned soils can be more vulnerable to erosion and invasive species colonization than unburned, particularly for severe burns occurring on steep slopes that eliminate vegetation, such as that in the western U.S. soils (Massman et al. 2003).

The absence of vegetation following severe fire may provide open space for weedy species to colonize. In a pinyon-juniper woodland, slash piles five years after burn had fewer plant species present, but four times more exotic species than outside slash pile burn areas (Haskins and Gehring 2004). Similarly, in mixed-species forests of the northwestern U.S., Scherer et al. (2000) found higher dominance of weedy species in areas in which logging slash had been piled and burned previously, compared to less disturbed areas.

Weedy or invasive species that have colonized severely burned patches in the western U.S. may profit from the presence, or in some cases, the absence of mycorrhizae. Exotic plant species found within both burned and unburned areas of pinyon-juniper woodland had significantly greater AM colonization than the established native plant species (Haskins and Gehring 2004). No clear differences in AM soil inoculum potential were found between burned and unburned sites, however, suggesting that either mycorrhizal abundance was not reduced by fire, or that mycorrhizae had recolonized the area. This study illustrates how the interactions between AM and weedy species may vary, producing different possible outcomes. Weedy species may do well if they are supported by mycorrhizal associates; oppositely, on a disturbed site where mycorrhizal populations have declined, weedy species that are non-mycorrhizal may grow better than mycorrhizal native species (Haskins and Gehring 2004).

Role of Revegetation in Changing the Soil Environment

Active revegetation of sites may help reduce the extent of nutrient loss and damage to soil structure (Certini 2005). Weston and Attiwill (1996) found that plant recolonization of Nfixing species after severe fires in *Eucalyptus* forest helped to compensate for N losses (Certini 2005). Physical changes in soil due to vegetation re-establishment can be produced by plant rooting which alters and strengthens soil physical structure. Tree root growth can aerate soil, decreasing compaction, and can increase ability of soil to retain water and nutrients (Fisher and Binkley 2000). Mycorrhizal hyphae associated with plants can extend through soil, stabilizing and strengthening soil structure (Jasper 1994). Soil porosity can also increase due to movement of soil microorganisms and their nematode and protozoan predators toward roots that release compounds appealing to microorganisms (Bardgett 2005).

The time frame over which the physical structure and the chemical composition of soil will begin to approximate more natural conditions of longleaf pine forest is likely to take decades at the least, although different soil components will change at different rates. A study by Markewitz et al. (2002) did not consider severe fire effects on soils, but did evaluate the agricultural legacy in longleaf pine stands of past soil disturbance and nutrient inputs. Longleaf pine stands from 1 to 14 years-old on formerly tilled soils were compared to natural stands on never-tilled soils. Bulk density, soil pH, and extractable NO_3^- and K were found to approach the values of never-tilled conditions at faster rates than cation exchange capacity (CEC), total C, N, P and extractable P, which advanced towards never-tilled conditions more slowly over time (Markewitz et al. 2002). Some inputs and alterations will have longer legacies than others.

Restoration of Other Ecosystems

Ponderosa pine forests share with longleaf pine forests structural and compositional features such as an open widely spaced overstory and grassy ground layer that promote cool-burning fires (Korb et al. 2004). Restoration efforts that thin trees to achieve more open tree cover produce a need for residual slash maintenance, such as through burning slash piles.

Consequent soil alteration of slash pile burn areas is not uniform. Likely due to spatial differences in heat intensity within a slash pile burn, fewer AM propagules and fewer viable seeds were found in the centers of fire scars, where heat was presumably the greatest during fire, compared to fire scar edges or areas outside of fire scars with more viable seed (Korb et al. 2004). With greater soil alteration, more time is needed for recovery, and thus time to natural revegetation generally takes longer in fire scar centers.

Korb et al.'s study also undertook revegetation treatments. After 11 months, areas that had received seed with living soil (i.e. including AM inoculants) as a treatment had more native vegetative cover (11.9%) than areas without revegetation treatments (0.01% native plant cover). In the second growing season, more plants flowered in the seed plus living soil treatment than in other treatments. Fewer exotic plants were found to establish in seed with living soil treatments as well.

Objectives:

The objectives of this study are to document soil changes resulting from burning large slash piles in the longleaf pine ecosystem, evaluate natural revegetation after burn, and examine the restoration potential of these severely burned areas through revegetation treatments. Specifically, this study will consider the following questions:

1. What are the effects of high intensity slash pile burns on soil and vegetation characteristics, and the soil seed bank?
2. How do these factors vary across soil types?
3. What is the rate of return of slash pile burn sites to pre-burn conditions?
4. What is the potential for re-establishment of longleaf pine and native ground cover species in fire scars, and do native topsoil amendments facilitate ground cover species establishment?

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CHAPTER 2

ALTERATION AND RECOVERY OF SLASH PILE BURN SITES IN THE LONGLEAF PINE ECOSYSTEM[1]

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ABSTRACT

Slash pile burns are used to eliminate unmerchantable wood debris that often results from forestry operations and which poses a future fire hazard. Slash pile burns have been used as part of longleaf pine restoration to remove encroaching hardwood species and initiate the return of frequent prescribed fire. When employed in a restoration context, the effects of slash pile burns may complicate restoration outcomes due to alterations to vegetation, soils and the soil seed bank. In this study, our objectives were to i) characterize pile fuel loads and temperatures above- and belowground during burn, ii) examine alterations to the soil seed bank, soil physical and chemical characteristics, and initial vegetation recolonization following burn across a soil moisture gradient, iii) determine the rate of return of soil and vegetation characteristics to pre-burn conditions, and iv) evaluate the potential for longleaf pine re-establishment in slash pile burn areas.

We found that slash pile burning of hardwoods results in elevated nutrient levels and significant impacts on vegetation and the soil seed bank. These effects remain evident for at least six years following burn. In this ecosystem, formerly weakly acidic soils become neutral to basic, and levels of P remain significantly higher. Following an initial decrease after burn, total soil N increases with time since burn. These changes suggest that not only does pile burning create a fire scar initially devoid of biota, but it produces a soil chemical environment very different from the surrounding area, which may have consequences for long-term longleaf pine ecosystem restoration efforts within landscapes that include numerous fire scars. Reintroduction of native pine species, as well as functionally important native ground cover species, would be recommended to overcome prescribed fire limitations that open the possibility for hardwood re-establishment in these areas.

INTRODUCTION

Slash pile burns are used to eliminate unmerchantable wood debris that often results from forestry operations and which poses a future fire hazard. Altered site conditions resulting from high fuel loads and long burn durations of wood piles are concentrated in localized areas, but may be abundant across the landscape. These pile burns often produce belowground temperatures that are lethal to soil biota as well as the soil seed bank (Roberts 1965; Korb et al. 2004; Massman and Frank 2004). Additionally, available nutrient levels are likely to be elevated for 1-2 years, or longer, following burn as a consequence of initial microbial death and ash deposition (Covington et al. 1991; Haskins and Gehring 2004; Korb et al. 2004; Jiménez Esquilín et al. 2007).

Slash pile burns are frequently implemented in restoration projects following tree thinning or selective removal of undesirable tree species (Korb et al. 2004; Phillips and Waldrop 2008). When employed in a restoration context, the effects of slash pile burns may complicate restoration outcomes if severe soil erosion occurs, or if undesirable species become established that impede natural ecosystem processes and functions. In the western U.S., bare spaces and altered soil conditions produced by slash pile burns have been demonstrated to serve as locations for non-native or ruderal species establishment (Haskins and Gehring 2004; Korb et al. 2004). As a result, these species may alter successional trajectories from that of the restoration objectives.

The legacy of slash pile burn fire scars may hinder the restoration process in the fire-maintained longleaf pine ecosystem, relative to the re-establishment of native herbaceous species necessary to promote frequent prescribed fire. Although burning of slash piles is commonly practiced after commercial pine plantation harvesting or with site preparation in the southeastern

U.S. (Carter and Foster 2004), the persistence of alterations to soil and vegetation characteristics is poorly known and implications in a restoration context have not been well-studied.

Restoration of *Pinus palustris* Mill. (longleaf pine) ecosystems that have become encroached with hardwoods often requires extensive hardwood removal to promote native herbaceous fine fuels needed to reinitiate regular prescribed fire (Williamson and Black 1981; Provencher et al. 2001; Kush et al. 2004). Given that many of the encroaching hardwood species (e.g. *Q. virginiana* Mill., *Q. nigra* L., *Q. laurifolia* Michx.) are not profitable in the current forest products market, large slash piles composed of whole trees of these unmarketable species may be burned. In the resulting fire scars, the presence of bare soil and persistence of less flammable ruderal species may hinder the spread of fire, thus creating sites for fire-intolerant hardwood re-establishment that perpetuates a cycle of woody encroachment (Waldrop et al. 1992; Jack and Conner 1999). Furthermore, it is uncertain whether native herbaceous savanna species would naturally recolonize fire scars, given that they have been found unlikely to re-establish even decades after physical soil disturbance (Hedman et al. 2000; Kirkman et al. 2004). Thus, fire scars may have long-lasting effects in these systems.

In this study we examined the effects of slash pile burns on soil and vegetation characteristics, and specifically considered their outcomes in the context of ongoing longleaf restoration projects. Specifically, we: i) characterized fuel load and quantified above- and belowground temperatures during burn, ii) examined alterations to the soil seed bank, soil physical and chemical characteristics, and initial vegetation recolonization following burn across a soil moisture gradient, iii) determined the rate of return of soil and vegetation characteristics to pre-burn conditions, and iv) evaluated the potential for longleaf pine re-establishment in slash pile burn areas immediately following the burn.

METHODS

Site description

All slash pile burn study sites were located on Ichauway (lat 31°15' N, long 84°30 W), an 11,700-ha private property of the Joseph W. Jones Ecological Research Center in the Lower Coastal Plain of southwestern Georgia (Baker County). The climate is humid sub-tropical with an average annual rainfall of 131 cm, dispersed evenly throughout the year. Temperatures range from 21-34°C in summer and 5-17°C in winter. The property is situated on karst topography and includes 9,700 hectares of second growth longleaf pine uplands having a diverse native ground cover including *Aristida stricta* Michx. (wiregrass), and 2,000 hectares comprised of depression-al wetlands, pine plantations, food plots and old fields. The sandy soils range from well- to poorly-drained. Much of the land was managed with frequent prescribed fire throughout the 20th century, typically in cool season months as part of quail management (Goebel et al. 2001). This burning pattern, together with landscape fragmentation, allowed hardwood seedlings to survive and grow into the canopy (Waldrop et al. 1992; Goolsby 2005). Prescribed fire intervals currently average 2 years (Goebel et al. 2001) across a range of seasons.

Slash piles of unmerchantable hardwoods (averaging 20 m in diameter and 4 m tall, in 2007 measures) are created on Ichauway as a result of a landscape-scale restoration effort to remove invasive fire intolerant hardwoods (e.g., *Quercus* spp.) and to reconnect a continuous fire corridor in the fragmented longleaf pine forest. Since 1993, over 3,600 ha. of the property has undergone hardwood removal, the majority since 2001. Estimated pile burn density in removal areas is approximately 1 pile burn per 3 ha., or 1,450 pile burns over the >3600 ha. area, based on surveys (covering 147 ha) within recent removal areas. During hardwood removal, trees are stacked into piles near the removal area using a hydro-axe, which causes limited soil disturbance.

Site Selection

To examine differences in fire impacts across a soil moisture gradient in 2007, we selected 16 unburned piles of hardwood slash on soils of two moisture regimes, for pre- and post-fire soil and vegetation sampling. We selected piles that were composed of 40-120 whole trees (based on counts of the number of tree butts exposed per pile). Eight piles were situated on well-drained upland soils (loamy sands, classified as fine-loamy, kaolinitic, thermic Typic Kandiudult) and eight on poorly-drained upland drain or depression soils (fine sandy loams, characterized as fine, kaolinitic, thermic Typic Paleaquult) (Soil Survey Staff, NRCS 2009). Henceforward, we will refer to the two soil moisture types as "mesic" (well-drained) and "hydric" (poorly-drained). The piles were burned six months following harvest. In 2008, we instrumented two additional slash pile burns at a separate mesic site to further characterize above- and belowground temperatures during burn. These piles were burned one month after construction.

To assess site recovery from the effects of slash pile burns, we established a chronosequence of fire scar locations by selecting 15 fire scars burned prior to 2007 (5 sites each from 2001, 2003 and 2006 burns). These previously-burned sites were located based on data from annual burn permit records, GIS maps of hardwood removal areas by year (dating back to 2001), and discussions with land managers, coupled with scouting for evidence of dark layers of ash and char in upper layers of soil. The practice of hardwood removal was more prevalent in mesic soil conditions during the period prior to 2007, and thus we restricted our search for older fire scars to mesic areas. Because the hardwood removal operations for a given year are spatially clustered, fire scars were located in different areas of Ichauway by burn year, rather than randomly spread throughout the area. We selected sites located on similar, well-drained soil types (loamy sands or sands, classified as Kandiudults).

Slash pile burn characterization

<u>Estimation of slash pile biomass</u>

Fuel loads of standing piles in 2007 and 2008 were estimated with tree butt counts and measurements of the exposed base diameter of each tree (Figure 2.1). To relate tree base diameter measurements to diameter at breast height (dbh), we used a linear regression for Southern hardwood species (Bylin 1982). We used a truck scale to weigh five entire trees (3 live oaks and 2 water oaks) ranging in dbh from 19-45 cm to determine the diameter-weight relationship using simple linear regression (PROC REG, SAS Institute Inc., version 9.1). From these estimated biomass and dbh measures, we estimated the total pile biomass.

<u>Aboveground temperatures during burn</u>

In July 2007, the initial study piles were burned (Figure 2.2). In May 2008, we burned the two additional sites. Aboveground temperatures during burn were monitored with the FLIR S60 digital thermal imaging system (Flir Systems, Inc, Boston, MA), which can register temperatures between -40°C to 1500°C, calculated by measuring radiation emitted from fire at wavelengths between 2-13 µm. We obtained thermal images of three burn events (18 July 2007, 19 July 2007, 28 May 2008). With a thermal imaging camera positioned 30 to 40 m from the slash pile, we recorded images continuously for 20 to 30 hours from the point of ignition, through the flaming stage, and to the smoldering stage. Following ignition, images were recorded at 5 sec intervals for the first hour, then reduced to intervals of 10 min, and 30 min as fire intensity declined. Thermal images were viewed and analyzed with ThermaCAM Researcher Pro software (FLIR

Systems, Inc.) to determine maximum aboveground temperatures over time and generate temperature-time response curves for the three burns.

Belowground temperatures during burn

Because the piles in 2007 were established prior to instrumentation, we attempted to measure belowground temperatures of the burn with HOBO data loggers (Onset Computer Corp., Bourne, MA) buried on the edges of 6 slash piles. Approximately 4-8 hours after ignition, we shifted burning logs directly above the locations of data loggers using a bulldozer. This approach yielded inconclusive results due to variation in fuel repositioning. Thus, to more precisely record belowground temperatures directly beneath burning piles, we installed data loggers at two mesic locations prior to slash pile construction and the subsequent burn in 2008. Only temperature results from the second year of burns are presented.

We used HOBO Type K (Chromel-Alumel) data loggers with a temperature sensor probe. The data logger ceases to perform at temperatures above 70°C, but the probe and attachment cord are rated to function at temperatures up to 1250°C. Thus, to protect the data logger from high temperatures, we buried them 1 m belowground along a 10 m long and 1 m wide trench at each of the two future slash pile sites. We inserted temperature sensor probes horizontally into the soil wall at one of four depths (5, 15, 30 and 50 cm). No probes were oriented directly above or below each other such that the soil column above each probe was undisturbed (Sackett and Haase 1992).

A total of 27 data loggers were buried between the two slash pile locations (13 beneath one pile, 14 beneath the other). Given the limited number of data loggers, 3 loggers were placed at 50 cm depth and all other depths had 8 data loggers divided equally between the two pile burns. Prior to installation, data loggers were placed inside plastic sealable containers with desiccant (Drierite Co. Ltd., Xenia, OH), and the temperature sensor cord was threaded through a drilled hole in the side of the plastic container. This hole was then sealed with silicone gel. The trench was refilled with soil after installation of the sensors. Harvested trees were then piled on top of the two instrumented sites and burned one month after installation, in May 2008. We programmed data loggers to obtain temperature readings continuously at 10 minute intervals and retrieved the instrumentation 1 week after burn. Data was uploaded to a personal computer using BoxCar ® Pro 4.3 (Onset Computer Corp.). Some data loggers below one of the two pile burns failed to retrieve data. Temperature data from both piles was combined to determine maximum temperatures, and average durations of lethal temperatures, for each soil depth.

Fuel and Soil Moisture

We measured fuel moisture of the hardwood piles on the mornings prior to ignition in both burn years. We cut a cross-section from the end of one tree within each slash pile under evaluation (16 piles total in 2007, and 2 piles in 2008). Each wood section was immediately wrapped in a paper bag, weighed, and dried in a 70°C oven until it reached a constant dry weight. Volumetric soil moisture (depth 30 cm) was measured at each site prior to burn using time domain reflectometry (TDR) (Evett 2003). Following burn, we placed flags along the edges of the fire scar to permit distinction of burned vs. unburned areas in future sampling.

Seed Bank Germination

We used a seedling germination technique (Poiani and Johnson 1988) to assess viability of the soil seed bank following fire. In July 2007, we collected six soil samples at each hydric and mesic pile location before and soon after the fire, for a total of 12 samples per pile, following a randomized complete block design that blocked by soil type and burn treatment. Soil samples collected pre-burn (controls) were located in the immediate vicinity of a hardwood pile, and soil samples collected post-burn were obtained within the burned area. Each soil sample consisted of a composite of five subsample soil cores (5.1 cm ht x 7.9 cm diameter) from within a 1-m^2 area. We stored soil samples at 4°C for 7 months prior to processing. In March 2008, we sieved each sample using a 2 mm screen to remove roots, and then spread forty percent of the soil (by weight) in plastic containers atop a 1:1 mixture of sand: potting soil mix.

Soil samples were positioned randomly in the greenhouse under natural light and were watered daily with deionized water. Emergent seedlings were grown until species identification could be made, and then discarded. Seedlings were identified on a biweekly basis. The study was concluded in October 2008 after all new seedling emergence had ceased.

Seedling recruitment rate per m^2 was determined as the mean of six subsamples per burn treatment (pre- or post-burn) at each pile. The number of species recruited per burn treatment at each pile was calculated as the sum of all species occurring in the six subsamples.

Soil Sampling

Chemical Samples

Soil samples for chemical analyses were collected in burned and adjacent unburned (control) areas at all fire scars in February-March 2008. We obtained composite soil samples with a soil probe at soil depths of 0-10 cm and 10-20 cm. In burned areas, we did not distinguish between the ash layer and beginning of the soil layer due to difficulty identifying the boundary. Soil samples within fire scars were taken along two perpendicular transects established across the fire scar and through the center (Figure 2.3). The first direction was chosen randomly. Subsamples were taken every 4 m along transects, beginning at 1 m inside the fire scar edge (and subsampling did not extend past 1 m inside the opposite end of the transect), and combined according to soil depth. Therefore, the number of subsamples varied with the size of the fire scar. Unburned soil samples were taken from four 8 m-long transects beginning on each side of the fire scar where the burned transect ended. A subsample was taken at 1 m, 5 m, and 9 m away from the fire scar edge on each of the four transects, and combined. Edges of fire scars for 2001 and 2003 pile burns were difficult to delineate precisely due to several years of vegetation regrowth that had occurred over time, and the fact that ash deposition is thinner on edges than toward the center of the pile. To ensure that control samples were located outside of the burn area, we established the sampling transect 5 m beyond the perceived edge of the fire scar (creating a buffer zone) and each transect extended 8 m from that point. Upon return to the lab, we removed a subsample of the composited soil sample to allow it to air-dry prior to pH analysis. Remaining samples were placed in a 65°C oven for 24 hours prior to total C, N, and P analyses. After oven drying, samples were further homogenized in 2 mm sieves and ground on a Spex 8000-D Mixer Mill to NIST standard soil texture. Total P was determined with a persulfate digestion (Nelson 1987) followed by colorimetric analysis on a Lachat Flow Injection Analyzer (Lachat Instruments, Milwaukee, Wisconsin, USA). The optional dilution step in the persulfate digestion was not followed because of the low P values typical of Coastal Plain soils. Total N and total C were determined with a micro-Dumas combustion assay using a CN Elemental Analyzer (ThermoElectron Corporation, Flash EA 1112 Series). Soil pH in water was measured in a solution of 20 g soil: 40 mL deionized water (Hendershot et al. 1993) using a standard pH meter. A subset of total P samples were analyzed at the University of Georgia Odum School of Ecology Analytical Chemistry Laboratory (Athens, GA). All other nutrient analyses were conducted at the J. W. Jones Ecological Research Center Analytical Laboratory (Newton, GA).

Physical samples

To determine soil bulk density of burned and unburned soils for each year of the chronosequence, soil cores were collected from all fire scar sites in November 2007-February 2008 with a double-cylinder, hammer-driven core sampler (Blake and Hartge 1986). At each fire scar, we obtained three samples inside the burn area (one sample each from center, intermediate, and edge locations) and 3 samples outside each fire scar. Soil samples were oven-dried at 105°C and then weighed to determine bulk density per unit volume (g/cm^3). The mean bulk density value per pile for each burn condition was determined. For soil texture analysis, we combined the three samples taken for bulk density analysis from each treatment area (burned or unburned) and homogenized it by passing twice through a 2 mm sieve. We used 100 grams of the combined soil for soil texture analysis with the hydrometer method (Bouyoucos 1936).

Vegetation Sampling

We sampled vegetation in August-October 2008, six months after soil sampling was completed. Vegetation in burned and adjacent unburned areas was sampled at five mesic 2007 fire scars and five hydric 2007 fire scars, as well as at all mesic fire scars burned in 2001, 2003, and 2006 (five each). Six 1-m^2 quadrats were sampled at each fire scar site, three in burned areas and three in adjacent unburned areas. At 2006 and 2007 fire scars, quadrats were located 5 m inside and 5 m outside of the fire scar edge, along 3 transects radiating from the fire scar center (Figure 2.4). Transects were 120 degrees apart. The first direction was chosen randomly. At piles burned in 2001 and 2003 where original edges were more difficult to ascertain, quadrats were located 2 m inside the conservatively determined edge and 10 m outside, to incorporate a buffer region that accounted for possible edge variation. In each quadrat, all plant species were identified and percent cover of each species was estimated visually with placement in cover classes. Species not identified in the field were collected, pressed and keyed. Nomenclature follows Wunderlin and Hansen (2003). Some species were combined by genus for analysis (i.e. *Andropogon spp., Chamaechrista spp., Galactia spp., Gamochaeta spp*., *Oxalis spp., Pityopsis spp.*), due to vegetative conditions at the time of sampling.

Species richness per m^2 was determined as the mean value of the three vegetation quadrats per burn treatment at each fire scar. Percent similarity (Jaccard) (Kent and Coker 1992) was calculated between each of the three paired burned and adjacent unburned quadrats per fire scar. Then a mean percent similarity per pile was obtained as the average of the three pairs. Each species was classed regarding growth habit, native or non-native status (USDA NRCS PLANTS Database), and its association with native undisturbed or disturbed site conditions (from species descriptions in Wunderlin and Hansen (2003)). Percent cover by class variables was calculated as the mean cover for all species per class for three quadrats in each burn treatment of each fire scar.

Pinus palustris Introductions

To consider pine survival in fire scars, container-grown longleaf pine seedlings were introduced in January 2008 to the eight mesic slash pile burn sites from the previous summer. Seedlings were planted only within fire scars, every 2 m along two perpendicular transects (Figure 2.5). Planting began at 1 m from the fire scar edge and ended at either 1 or 2 m from the opposite edge, depending on fire scar size. The number of pines planted per fire scar was dependent on the fire scar size, and a total of 168 seedlings were planted. Pines received no artificial watering. Survivorship was observed in June 2008, October 2008 and March 2009 (4, 8, and 13 months after planting, respectively). In the final census, each seedling was also categorized based on its location along the burn gradient into a fire scar zone: “Edge” (1-3 m from fire scar edge), “Intermediate” (4-7 m from fire scar edge), or “Center” (8 m to center point [m] from fire scar edge).

Statistical Analyses

We used paired t-tests to determine significant differences between paired burned and unburned values associated with fire scar locations within each burn year, for soils and vegetation data (PROC MEANS, options t and prt, SAS Institute Inc., version 9.1). To make comparisons among burn years, we used a difference term between burned and unburned conditions as a response variable. Difference terms were also used in linear regression analysis of soils and vegetation data to consider changes with time since burn (PROC REG, SAS Institute Inc., version 9.1). The approach of using difference terms accommodated the often heteroscedastic nature of data owing to the greater variance associated with burned characteristics in contrast to unburned values, and was appropriate given the paired sampling of burned and adjacent unburned areas at all sites.

To determine effects of pile burning on the soil seed bank between soil types, we determined the difference between pre- and post-burn soil seed bank values per pile and then used t-tests to examine mean differences of this variable among soil types (PROC TTEST, SAS Institute Inc., version 9.1).

Species richness and percent cover were analyzed as above for within and between year comparisons. Mean Jaccard vegetation similarity values between burned and adjacent unburned fire scar locations over the chronosequence were rank-transformed due to heterogeneity of variance and entered into analysis of variance for comparisons among burn years (PROC ANOVA, SAS Institute Inc., version 9.1), and in linear regression to consider changes over time (PROC REG, SAS Institute Inc., version 9.1). Means and standard errors before transformation are reported.

Pine seedling survivorship at 13 months after planting was evaluated with ordered logistic regression (PROC LOGISTIC, SAS Institute Inc., version 9.1), with the variable of seedling planting zone (edge, intermediate, center) included to determine if survivorship varied along the burn gradient.

RESULTS

Temperatures During Burn

Average fuel loads were greater at the hydric site (75 ± 11 Mg/pile) than the mesic site (36 ± 6 Mg/pile), a consequence of more trees per pile as well as higher average tree diameters (Table 2.1). These two piles had estimated fuel loads of 77 and 96 Mg/pile, respectively.

Aboveground temperatures exceeded 1000°C soon after ignition, and flaming or smoldering continued for at least 24 hours at all pile burns monitored in 2007 and 2008 (Figure 2.6).

Maximum and average belowground temperatures as well as durations by soil depth are listed in Table 2.2. Lethal belowground temperatures (>60°C) persisted for more than 3 days at soil depths of 5 and 15 cm, and approximately 2 days at 30 cm below one pile burn (Figure 2.7). Additionally, soils in the center areas of fire scars demonstrated the characteristic soil reddening associated with near total removal of organic matter and transformation of Fe-oxides, with a blackened charred soil layer immediately below (Ulery and Graham 1993; Ketterings and Bigham 2000).

Soil moisture values measured on the days of burn in 2007, within a period of drought, did not differ between the two soil types ($F=4.32$, $p<0.06$) (Table 2.3). Soil moisture measurements for burns in 2008 had similar soil moisture levels at the time of burn as with burns in the previous year.

Fuel moisture values did not differ between hydric and mesic piles in 2007, but the two 2008 piles had significantly greater fuel moisture than the previous year ($F=64.5$, $p<0.0001$; contrast “2007 vs 2008”, $F=128.95$, $p<0.0001$) (Table 2.4). The two 2008 piles also had higher fuel loads, but the burn duration was similar.

Seed Bank

Soil seed banks of mesic and hydric soils differed prior to burn in number of seedlings ($F=9.23$, $p<0.01$) and number of species ($F=17.01$, $p=0.001$), with mesic soils higher in both respects, but regardless of soil type, both experienced similarly low seedling recruitment rates (<500 per m^2) in burned soils ($F=0.01$, $p=0.90$), and similarly low species richness ($F=0.13$, $p=0.70$) (Table 2.5).

Soils

Data from 0-10 cm soil depth are reported throughout, and in the few cases where differences between burned and unburned soils were found in the 10-20 cm layer, they are reported as well. Among unburned sites, mesic soils had higher total P than hydric soils ($F=11.49$, $p<0.01$) (Table 2.6), and after burn mesic soils had higher total P as well ($F=5.75$, p<0.05). Similarly, mesic sites had higher pH relative to hydric sites in burned soils (F=6.91, p<0.05). Yet hydric and mesic soils did not differ for any soil nutrient when paired difference values (burned-unburned) were analyzed, suggesting that both underwent similar net changes from burn, despite differences in fuel load.

In the 0-10 cm soil layer, total P was greater in fire scars than adjacent unburned areas at all years in the chronosequence (Table 2.7). Mean levels of P increased by 4 times the unburned levels in the year after burn, and the difference between burned and unburned decreased over time. However, even after 6 years, levels of total P in burned sites remained 2.4 times greater than unburned levels (Figure 2.8).

Similarly, soil pH in the top 10 cm of burned sites was elevated relative to unburned sites for all years (Table 2.7), with differences decreasing over time (Figure 2.9). This pattern was also apparent in the 10-20 cm layer for mesic soils in all burn years (p<0.001) (Table 2.8), as well as in 2007 hydric soils (burned pH: mean = 6.7 ± 0.2; unburned pH: mean = 5.3 ± 0.1, t = 18.18, p<0.0001).

The mean paired differences of percent carbon for burned and unburned conditions did not differ between years (Figure 2.10). Percent total nitrogen was lower in burned areas relative to unburned areas in the year after burn (Table 2.7). However, over time, percent nitrogen increased in burned sites, and by Year 4 after burn, burned values exceeded that of unburned areas in the 0-10 cm soil layer (Figure 2.11).

Paired differences in soil bulk density for burned and unburned areas differed only in Year 6 (t value = -3.34, p<0.05) (Table 2.9), with burned areas having lower bulk density values. Soil textures were classified as sands or loamy sands, and burned areas sometimes showed increased silt and clay fractions relative to unburned soils.

Vegetation Re-Establishment

Vegetation species richness was reduced in fire scars compared to adjacent unburned areas in all years except Year 7 in the vegetation chronosequence ($p<0.01$) (Table 2.10), with differences between burned and unburned areas lessening over time (Figure 2.12). Accordingly, percent similarity of burned and unburned vegetation increased with time since burn, ranging from 7% in Year 2 to 22% in Year 7 after burn (Figure 2.13).

Species richness of native species restricted to undisturbed areas (Table 2.11) (Wunderlin and Hansen 2003) was lower in burned sites than unburned in all years after burn ($p<0.05$) (Table 2.12). Native species richness showed no trend of increasing with time after burn beyond the first year, and the difference term between burned and unburned species richness did not significantly differ between years ($p>0.70$). Percent cover of native species was significantly lower in burned areas in Years 1 and 5 after burn ($p<0.05$) (Figure 2.14). Trends were not as clear in other categories of the disturbance classification, although undesirable native species (composed of woody species and the shrub *Rubus cuneifolius* Pursh.) were initially eradicated in burned areas in the first year after burn, but began to recolonize with time.

Percent cover by the Asteraceae family (composites) was significantly greater in burned areas compared to unburned areas at all years after burn (Figure 2.15a). In unburned areas, percent cover of Asteraceae averaged between 8.3 ± 2.2% and 15.8 ± 6.7% in all years, whereas in burned areas it varied between a maximum of 47.0 ± 6.3% cover in Year 2, to 30.9 ± 2.2% in Year 7. Certain ubiquitous species of this family showed dominance of fire scars in certain years. *Conyza canadensis* L. was found at low levels at all sites, but rose to 23.5 ± 4.9% cover in fire scars one year after burn (2007), and dropped to below 2% in all other chronosequence years. *Eupatorium capillifolium* (Lam.) Small was found in fire scars at all times since burn, but reached 28.7 ± 10.5% cover in fire scars at 2 years after burn, and varied between 0.8 and 7% cover in fire scars in other years. Lastly, *Solidago canadensis* L. reached 17.2 ± 5.8% cover in fire scars at 5 years after burn (2003), and 9.2 ± 2.4% cover at 7 years after burn (2001).

Poaceae (grasses), another important family, had significantly less cover in burned areas in Year 2 after burn (Figure 2.15b), and in all other years demonstrated an insignificant trend in reduced cover in burned areas compared to unburned areas. Other families of interest, such as Fabaceae (legumes) and Fagaceae (oaks) did not show clear or significant trends over time. Occurrence of Fabaceae was variable in unburned areas, and in its limited recolonization of fire scars, varied from 0.7 ± 0.3 % cover in Year 1 after burn, to 3.4 ± 3.1 % cover in Year 6. Fagaceae varied from 0% coverage in the year after burn, to 2.3 ± 2.0 % cover in Year 7.

Analyses by growth habit did not reveal conclusive results and thus are not presented. Additionally, non-native species were rarely found in fire scars or in the surrounding area, and were not further analyzed.

Pinus palustris Introductions

Percent survivorship of longleaf seedlings planted in 8 mesic fire scars (168 seedlings planted total) was 98% at four months after planting in early June 2008. At 8 and 13 months after planting, survivorship was constant at 70.9 ± 3.5%. Seedling survivorship increased along the burn gradient, with 58% survivorship in the edge zone (1-3 m from fire scar edge), 76% survivorship in the intermediate zone (4-7 m from fire scar edge), and 80% survivorship in the center zone (8 m to center point [m] from fire scar edge) (Figure 2.16). A significant logistic regression was developed using fire scar zone (zone ordered as Edge=1, Intermediate=2, Center=3), which predicted increased pine survivorship along the burn gradient: log (odds of survivorship) = -0.1702 + 0.5640*zone. The goodness of fit for this regression was $p>0.44$, indicating that the model should not be rejected.

DISCUSSION

This study demonstrates that slash pile burning of hardwoods within a nutrient-poor ecosystem results in elevated nutrient levels and significant impacts on vegetation and the soil seed bank that remain evident for at least six years following burn. In this ecosystem, formerly weakly acidic soils become neutral to basic, and levels of phosphorus remain significantly higher. Despite evidence indicating that N volatilization occurs at the time of burn, total N increased with time since burn. These changes suggest that not only does pile burning create a fire scar devoid of biota after burn, but it produces a soil chemical environment very different from the surrounding area, which may have consequences for long-term longleaf pine ecosystem restoration efforts within landscapes that include numerous fire scars.

Fire Temperatures

Soil temperatures that cause instantaneous plant tissue death (>60°C, Wade and Johansen 1986) were measured in this study to 0.5 meter belowground, the consequence of high temperatures produced over a long combustion period. The extended duration of lethal soil temperatures >60°C (averaging between 47 and 76 hours at all soil depths) would be likely to eradicate most other soil biota as well (DeBano et al. 1998). Belowground temperatures are within the ranges documented in other slash pile burns (Roberts 1965; Massman et al. 2003). Total vaporization of water likely occurred to at least 15 cm belowground (>95°C, Certini 2005; >100°C, Raison 1979), as temperatures cannot increase above 100°C until water has evaporated

(Raison 1979). Given the depth and duration of elevated temperatures, it is not surprising that the soil seed bank was effectively eradicated by burn, even though some seed bank species are known to have thick seed coats (e.g. Fabacaeae) that may germinate in response to much briefer exposures to severe temperatures (Baskin and Baskin 1998; Herranz et al. 1998). The similarly low recruitment in the post-burn seed bank soils further suggests that both hydric and mesic soil types experienced similarly severe conditions during burn. The severely reduced recruitment from the soil seed bank following burning in this study is consistent with reports of soil sterilization by Korb et al. (2004) and has consequences for recolonization post-fire. With vegetation in the burned area destroyed and the seed bank effectively eradicated, plant recolonization of fire scars is dependent on seed dispersal from outside the pile.

Soils

The increase in total C in some years of the chronosequence is consistent with other high severity fires (Neal et al. 1965; Johnson and Curtis 2001), and is likely due to char inputs and translocation of organic substances into soil below the surface (DeBano et al. 1976; Johnson and Curtis 2001). Little change in carbon levels between years following burn would be expected, and the lower carbon levels in the first years after burn in fire scars compared to later years could be an artifact of intentionally not including surficial charred wood and black carbon in samples. In later years, the char fragments would be more likely to settle into soil and consequently be sampled (Johnson and Curtis 2001; DeLuca and Aplet 2008). Some black carbon may also have been removed during soil sieving as part of sample preparation (Neal et al. 1965). An increase in soil carbon due to char has implications for soil-water relations and soil microorganisms such as higher moisture retention capacity, higher capacity for ion adsorption, and the provision of habitat for microorganisms such as bacteria and fungi (Tryon 1948; DeBano et al 1998; Pietikäinen 1999; DeLuca and Aplet 2008).

The loss of total soil nitrogen as a consequence of burn was expected based on high surface and belowground temperatures (Raison et al. 1984; Little and Klock 1985; Feller 1988; Korb et al. 2004). Our observation of a pattern of increasing N with time since burn is also consistent with other studies (Gifford 1981; Covington et al. 1991; Smithwick et al. 2005), and is likely attributable to multiple factors. One partial explanation is that N may have volatilized and followed temperature gradients downward into the soil, to condense at a cooler soil layer (DeBano et al. 1976).

Another explanation for the initial N decrease and subsequent increase may be found in the lack of surface char incorporated into soil samples in the year after burn. Char may have stored N that that serves as a nitrogen input, countering volatilization losses from soil (Johnson and Curtis 2001). Further, some evidence exists suggesting that hardwood char may have higher levels of N than conifer char (Tryon 1948).

Additionally, N fixation activity can be heightened in environments having warm and basic conditions (Smithwick et al. 2005), a characteristic typical of fire scars due to the initial deposit of dark char and absence of vegetation for several months (Raison 1979). Bacterial numbers have been found to increase after slash pile burns and comprise distinctly different communities than adjacent unburned areas (Jiménez Esquilín et al. 2007); these bacterial communities may include asymbiotic N-fixers (Smithwick et al. 2005; Janzin and Tobin-Janzin 2008), although their role is debated (Binkley et al. 2000). As plant recolonization proceeds, symbiotic fixation may begin to occur as legumes recolonize burned areas (Johnson and Curtis

2001). In our study, however, we found only small recolonization gains by legumes over the chronosequence.

Although potential inputs of N from precipitation and dry deposition would be low (2.8-5.9 kg/ha/yr) (Carter and Foster 2004), these sources may have contributed to increases in N if taken up by microbes in fire scars or adsorbed to char. Given the greater moisture retention and adsorption capacity of char relative to mineral soil (Tryon 1948; Zackrisson et al. 1996; Pietikäinen 1999; DeLuca and Aplet 2008), rainfall nutrient inputs could likely be retained.

The large and persistent increases in total P and pH after burn are highly associated with hardwood combustion that frees available forms of P and basic cations (Ca, Mg, K) contained in the deposited ash (Raison et al. 1985; Feller 1988; Ohno and Erich 1990; Khanna et al. 1994; DeBano et al. 1998). Additionally, some increases in available P and basic cations would result from soil heating and the break down of organic matter in soils (Chambers and Attiwill 1994; DeBano et al. 1998). Although our study did not measure available P, many studies have documented significant increases in P availability in the years following burn (Humphreys and Lambert 1965; Haskins and Gehring 2004; Jiménez Esquilín et al. 2007). Yet other studies of pile burns have found increases in P to be more fleeting (Gifford 1981) or for total P, not significant (Korb et al. 2004), which may be due to smaller fuel loads in those studies (Feller 1988). Alternatively, in soils extremely low in P (such as our study site), the increases in total concentration may be more striking relative to that of findings in locations with naturally higher levels of soil P (e.g. Korb et al. 2004).

In basic conditions, P bonds with Ca and thus can be made unavailable for biotic uptake (Kwari and Batey 1991). As soil becomes more neutral over time with leaching of basic cations, P would be likely to become more available (DeBano et al. 1998; Fisher and Binkley 2000).

Given the mildly basic soil conditions found after burn in this study, some P limitation may have occurred initially despite the total P increase. Phosphorus would be expected to remain immobilized in soils rather than to leach out of the system (DeBano et al. 1998), and we found no evidence of P increases in the 10-20 cm layer of fire scars across the chronosequence to suggest that translocation was occurring.

Soil pH increased from weakly acidic to weakly alkaline levels within the first year following burn, and at six years after burn was still elevated to neutral levels. In other slash pile burn studies, pH has been found to remain elevated at 1 year after burn (Neal et al. 1965), 15 months (Jiménez Esquilín et al. 2007), 2 years (Arocena and Opio 2003), and 3 years after burn (Jönsson and Nihlgard 2004), while in some cases no significant effect was found (Gifford 1981). For pH, the presence of carbonates in soils can buffer pH response (Certini 2005). Sandy soils in the longleaf system are low in carbonates (Soil Survey Staff, NRCS 2009), which may partly explain the strong pH increase after burn.

The greatest decline in levels of P and pH over the chronosequence occurred between 0.5 and 1 year after burn, likely due to wind-dispersed losses of fine ash particulates prior to natural revegetation establishment. Erosion of ash from fire scars would be minimal in the low-lying Coastal Plain landscape, compared to steeper landscapes in which erosion results in the bare conditions that follow severe fires, removing levels of P found in upper soil layers (Giardina and Rhoades 2001). The elevated soil pH found in the 10-20 cm soil layers of fire scars suggests translocation and leaching of some basic cations is occurring over the soil profile, as expected (DeBano et al. 1998).

A more natural example of severe but localized fires that can occur in this system are found in log burn-outs, typically of fallen pine trees. We sampled soils from five log burn-outs in the longleaf system to evaluate possible phosphorus inputs found in those locations to compare to the elevated levels found with hardwood burning. Log burn-outs may cause lethal temperatures directly below logs (Monsanto and Agee 2008) and soil sterilization (Rhoades et al. 2004), but based on our limited sampling with no differences found between burned and unburned areas (t value = 1.86, $p>0.13$), we conclude that they do not result in elevated P to the extent found from hardwood slash pile burning. This result is clearly a consequence of differences in fuel load, but may also be related to differences in nutrient content by wood species (Hakkila 1989; Pitman 2006), and underlines the alteration due to slash pile burns that is outside the normal bounds of localized high fire intensity in this system.

The general lack of change in soil bulk density as a consequence of burn has been noted in other studies (Massman and Frank 2004), and a change would be more expected to occur in soils having higher clay contents, which when exposed to high temperatures would lead to aggregation and coarsening of soil texture (Chambers and Attiwill 1994). Coastal plain soils with high sand percentages would be less likely to undergo coarsening, as was the case in this study.

Vegetation Re-establishment

In contrast to reports documenting increased colonization by non-native plant species (Harrod and Reichard 2002; Haskins and Gehring 2004; Korb et al. 2004), the near absence of non-native species in our study likely reflects the fact that relatively few non-native species are present in sites adjacent to our study areas.

Lower species richness in fire scars relative to unburned sites in the first 5 years after burn, coupled with the low but increasing levels of vegetation similarity between burned and adjacent unburned areas with time indicate the significantly altered vegetation community that results in fire scars. The steady increase in percent similarity over time between burned and unburned areas suggests that some contribution of propagules or vegetative expansion into fire scars is occurring from species in adjacent unburned areas. However, even after 6 years, the percent similarity between burned and unburned sites remains less than that expected between plots of undisturbed longleaf pine-wiregrass savanna (Kirkman et al. 2001).

The low number of native species characteristic of undisturbed sites (Wunderlin and Hansen 2003) that returned to fire scars is undoubtedly due in part to the eradicated seed bank resulting from burn, but also likely a result of limitations in the dispersal abilities (Hedman et al. 2000; Kirkman et al. 2004) and possibly germination requirements (Izhaki et al. 2000) of these species.

The persistent dominance by Asteraceae and diminished cover of Poaceae in fire scars is important in that the absence of fine fuel may create safe sites from fire for possible hardwood establishment and growth over time (Mitchell et al. 2006), particularly given elevated pH.

Pinus palustris Introductions

The high survivorship 1 year following burning and planting suggests that longleaf pine establishment is possible in fire scars, despite elevated pH and P. The increased pine survivorship in more interior areas of fire scars may be due to progressively reduced competition from recolonizing plants in those areas. Although pines are not usually associated with basic soils, introductions of other pine species into pile-burned areas have proven successful. Growth of *Pinus taeda* (loblolly pine) after three growing seasons was greater when planted into burned windrow areas compared to outside of them (Applequist 1960), and where *Pinus rigida* was planted into an area after clearcut and debris-burning, growth of pines in ashbed areas 9 years after planting was greater than in adjacent areas, with growth correlated with increased levels of available P (Humphreys and Lambert 1965). Although longleaf pines may establish and grow in fire scars, they may be more sensitive to disease and insect problems, however (M. Hainds, personal communication).

CONCLUSIONS

In this study, the extremely high fuel loads of slash piles resulted in lethal belowground temperatures to 50 cm soil depth for several days when burned. The high temperature and duration of the fire nearly eliminated the soil seed bank in both hydric and mesic soil types. Regardless of soil type, patterns of change in soil chemical properties associated with the burn were similar. Furthermore, nutrient alterations as a result of fire persisted over time. Elevated soil nutrients, including total P and pH, persisted in fire scars across the six-year mesic chronosequence of time since burn. Total N decreased immediately after burn, but over the six-year interval increased to levels above that of unburned conditions. Soil texture and bulk density were not significantly altered by burning. The species richness and cover of native species associated with undisturbed areas were reduced in fire scars after burn, suggesting that these species may have dispersal limitations to recolonization. Total grass cover was reduced and ruderal species of the Asteraceae dominated after burn. The poor fuel quality of these species, coupled with bare ground are likely to result in prescribed burns that are patchy and insufficient to top-kill hardwood seedlings that will inevitably invade fire scars in the absence of fire.

Although overall survivorship of planted pine seedlings after one year was 70%, greater survivorship occurred in more interior zones of fire scars relative to edge zones, perhaps due to reduced competition effects where vegetation was slower to recolonize.

Our study identifies alterations in the biota and soil characteristics after burn that have immediate effects on vegetation re-establishment, and the persistence of these altered conditions may have ramifications for meeting restoration objectives of the longleaf pine ecosystem. These findings are particularly relevant to this fire-maintained ecosystem because of the dominance by ruderal, less flammable species that are likely to promote re-establishment of hardwood species in the absence of fuels to carry prescribed fire. We demonstrated that successful establishment of longleaf pine seedlings within the fire scar site is possible immediately following burn and may offer potential management opportunities for re-connecting fire corridors. Particularly if coupled with introduction of native grasses, the needle cast from planted longleaf pine seedlings species should prove helpful in advancing the successional trajectory of the restoration site.

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Table 2.1 Estimated fuel loads of slash piles (standard errors in parentheses)

Burn year	Slash pile	Number tree butts	Estimated average tree dbh (cm)	Mg/pile	Mean Mg/pile
2007	mesic1	68	23	30.5	35.7 (4.6)
2007	mesic2	72	23	32.0	
2007	mesic3	60	22	21.0	
2007	mesic4	110	24	53.0	
2007	mesic5	62	25	35.0	
2007	mesic6	42	26	24.9	
2007	mesic7	63	24	30.9	
2007	mesic8	83	27	58.0	
2007	hydric1	63	30	55.0	74.8 (10.0)
2007	hydric2	94	29	73.6	
2007	hydric3	103	29	83.8	
2007	hydric4	60	33	62.8	
2007	hydric5	59	30	51.9	
2007	hydric6	142	31	135.6	
2007	hydric7	98	32	97.8	
2007	hydric8	100	22	38.1	
2008	mesic9	193	23	76.8	86.2 (9.3)
2008	mesic10	214	23	95.5	

Table 2.2 Maximum belowground temperatures and mean duration of lethal temperatures (>60 °C) associated with two slash pile burns (standard errors in parentheses).

Soil depth (cm)	N (data loggers)	Max. temp. (°C)	Mean max. temp. (°C)	Mean duration >60°C (hrs)
5	5	288	184 (32)	76 (9)
15	6	173	119 (12)	76 (7)
30	7	93	83 (3)	59 (9)
50	2	71	71 (1)	47 (10)

Table 2.3 Mean volumetric soil moisture by soil type on days of burn (standard errors in parentheses).

Burn year	N (slash piles)	Soil type	vsm (%)
2007	8	hydric	5.35 (0.72)
2007	8	mesic	9.01 (1.61)
2008	2	mesic	7.37 (1.19)

Table 2.4 Mean percent fuel moisture by soil type on days of burn (standard errors in parentheses).

Burn Year	N (slash piles)	Soil type	Fuel moisture (%)	Months since construction
2007	8	hydric	16.19 (1.67)	>6
2007	8	mesic	16.62 (0.85)	>6
2008	2	mesic	50.31 (4.51)	1

Table 2.5 Seed bank germination by soil type and burn status (standard errors in parentheses). * indicates a significant difference between burned and unburned values by soil type. Pre-burn values significantly differ between soil types in both number of seedlings and number of species ($p<0.01$) but post-burn values do not ($p=0.94$).

Soil type	Burn status	Seedlings/m^2	Mean species/pile	Total species number (87 total)
Mesic	pre-burn	12733 (1975)*	28.9 (1.2)*	64
	post-burn	442 (301)*	3.5 (1.0)*	16
Hydric	pre-burn	5589 (1277)*	21.1(1.5)*	43
	post-burn	471 (266)*	4.0 (0.9)*	16

Table 2.6 Soil chemical properties from hydric and mesic 2007 piles, with burned-unburned difference terms included (standard errors in parentheses). Difference terms are not significantly different by soil type for any chemistry variable, although unburned mesic soils are significantly higher in total P than unburned hydric soils, and levels of P and pH in burned soils are significantly greater in mesic soils than hydric soils. * indicates that a burned and unburned pair (for respective nutrient and year) significantly differ by paired t-test (p<0.05).

Soil type and burn year	Burn status	Total C (%)	Total N (%)	Total P (%)	pH
Hydric 2007	Burned	1.814 (0.136)	0.058 (0.006)	0.034 (0.006)*	7.7 (0.1)*
	Unburned	1.818 (0.096)	0.068 (0.006)	0.008 (0.001)*	5.5 (0.1)*
	B – U	- 0.004 (0.083)	- 0.01 (0.006)	+ 0.026 (0.005)	+ 2.1 (0.2)
Mesic 2007	Burned	1.577 (0.159)	0.043 (0.008)*	0.052 (0.005)*	8.3 (0.2)*
	Unburned	1.522 (0.111)	0.057 (0.006)*	0.013 (0.001)*	5.7 (0.1)*
	B – U	+ 0.055 (0.094)	- 0.014 (0.003)	+ 0.039 (0.005)	+ 2.5 (0.2)

Table 2.7 Soil chemical properties (0-10 cm) from across mesic chronosequence (standard errors in parentheses). * indicates that a burned and unburned pair (for respective nutrient and year) significantly differ by paired t-test ($p<0.05$). One value for P (%) from 6 years since burn was removed from analysis due to likely lab error, in which P was measured as 3 times higher than any other site in any other year.

Years since burn	Burn status	C (%)	N (%)	P (%)	pH
0.5	burned	1.577 (0.159)	0.043 (0.008)*	0.052 (0.005)*	8.3 (0.2)*
	unburned	1.522 (0.111)	0.057 (0.006)*	0.013 (0.001)*	5.7 (0.1)*
1	burned	2.165 (0.239)	0.062 (0.010)	0.035 (0.006)*	7.7 (0.1)*
	unburned	1.690 (0.116)	0.050 (0.004)	0.010 (0.001)*	6.0 (0.1)*
4	burned	2.198 (0.297)	0.072 (0.012)*	0.031 (0.003)*	7.4 (0.1)*
	unburned	1.531 (0.134)	0.048 (0.008)*	0.013 (0.002)*	5.7 (0.1)*
6	burned	2.182 (0.287)	0.092 (0.010)	0.029 (0.005)*	7.6 (0.2)*
	unburned	1.406 (0.105)	0.060 (0.005)	0.012 (0.001)*	6.0 (0.1)*

Table 2.8 Soil pH (10-20 cm) across mesic chronosequence (standard errors in parentheses). * indicates burned and unburned values paired by year significantly differ ($p<0.05$). Difference terms (B-U) do not significantly differ across burn years ($p>0.09$).

Time since burn (years)	Burned (pH)	Unburned (pH)
0.5	6.8 (0.2)*	5.3 (0.1)*
1	6.6 (0.2)*	5.5 (0.1)*
4	6.3 (0.2)*	5.5 (0.1)*
6	6.7 (0.2)*	5.5 (0.1)*

Table 2.9 Soil bulk density values of burned and unburned areas by time since burn (standard errors in parentheses). * indicates significantly different values within a given year ($p<0.05$).

Time since burn (years)	Burned (g/cm^3)	Unburned (g/cm^3)
0.5	1.48 (0.05)	1.49 (0.04)
1	1.39 (0.08)	1.35 (0.04)
4	1.26 (0.08)	1.39 (0.03)
6	1.19 (0.11)*	1.43 (0.05)*

Table 2.10 Species richness across mesic chronosequence (standard errors in parentheses). * indicates burned and unburned values significantly differ within a given year ($p<0.05$).

Time since burn (years)	Burned (No. species / m^2)	Unburned (No. species / m^2)
1	4.93 (0.93)*	12.73 (1.30)*
2	6.07 (0.65)*	10.40 (1.06)*
5	9.13 (0.67)*	12.93 (0.63)*
7	9.53 (0.87)	11.53 (0.39)

Table 2.11 Disturbance classification from Wunderlin and Hansen (2003) based on species accounts. The category “undesirable native” was created to describe restoration objectives.

Disturbance classification (adapted and **modified from Wunderlin and Hansen 2003)	
both	Found in both disturbed and native undisturbed areas
disturbed	Found only in disturbed areas
native	Restricted to native undisturbed areas
**undesirable native	Species classified as native that inhibit prescribed fire movement, including woody species (such as *Quercus spp*.) and the shrub *Rubus cuneifolius*

Table 2.12 “Native” species richness between burned and adjacent unburned areas over chronosequence (standard errors in parentheses). * indicates burned and unburned values significantly differ within a given year ($p<0.05$).

Years since burn	Burned (No. native species / m^2)	Unburned (No. native species / m^2)
1	0.90 (0.24)*	4.00 (0.55)*
2	1.80 (0.36)*	4.40 (0.41)*
5	2.00 (0.35)*	4.80 (0.90)*
7	1.80 (0.27)*	3.87 (0.47)*

Figure 2.1 Image of slash pile (“Mesic 8”) before burn. White stake standing perpendicular to pile is 5 m tall.

Figure 2.2 Image of slash pile (“Hydric 2”) after burn. Tripod in front of pile is 1.5 m tall.

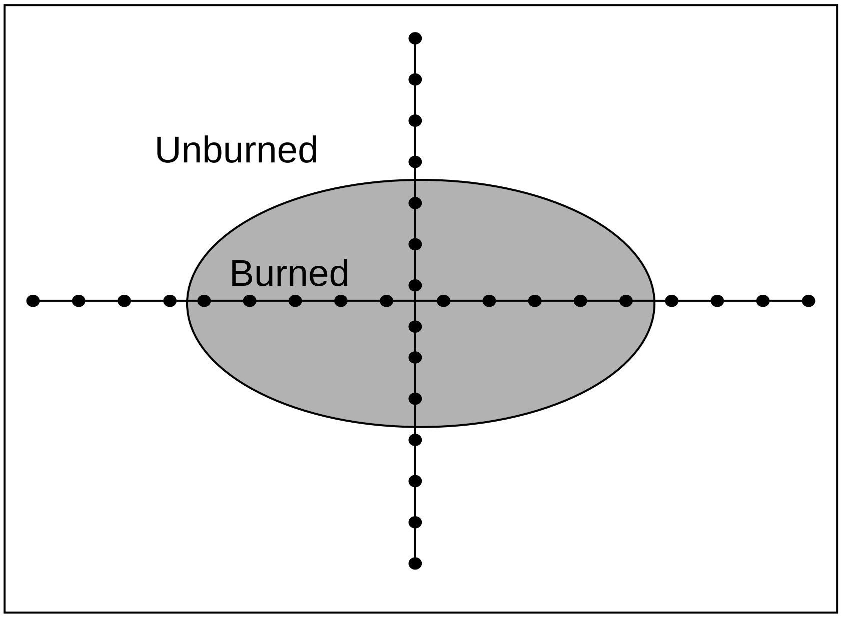


Figure 2.3 Sampling design for soil chemistry analyses for each fire scar. Subsamples were obtained along two perpendicular transects across the fire scar site and composited according to burn status.

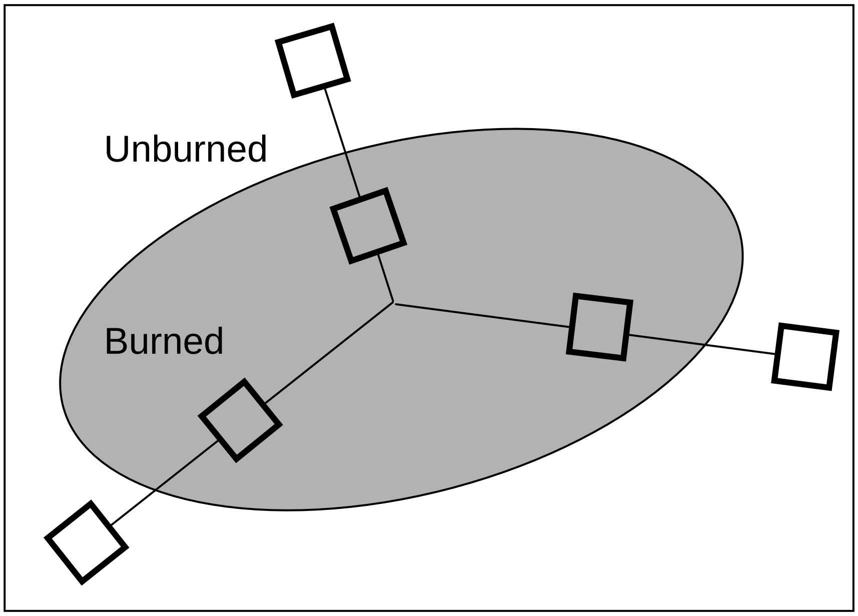


Figure 2.4 Vegetation sampling design with paired burned and unburned quadrats (1-m^2) located along three transects extending from fire scar center.

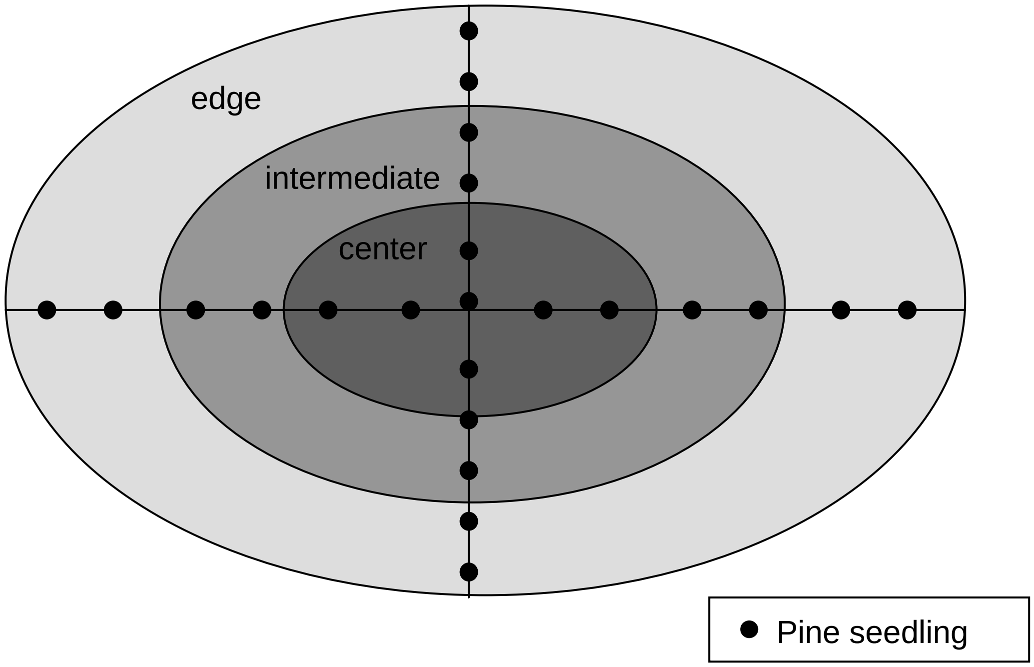


Figure 2.5 Longleaf pine seedling planting design in fire scars. Pine seedlings were planted at 2 m apart along two perpendicular transects across the burn gradient (edge, intermediate, center zones)

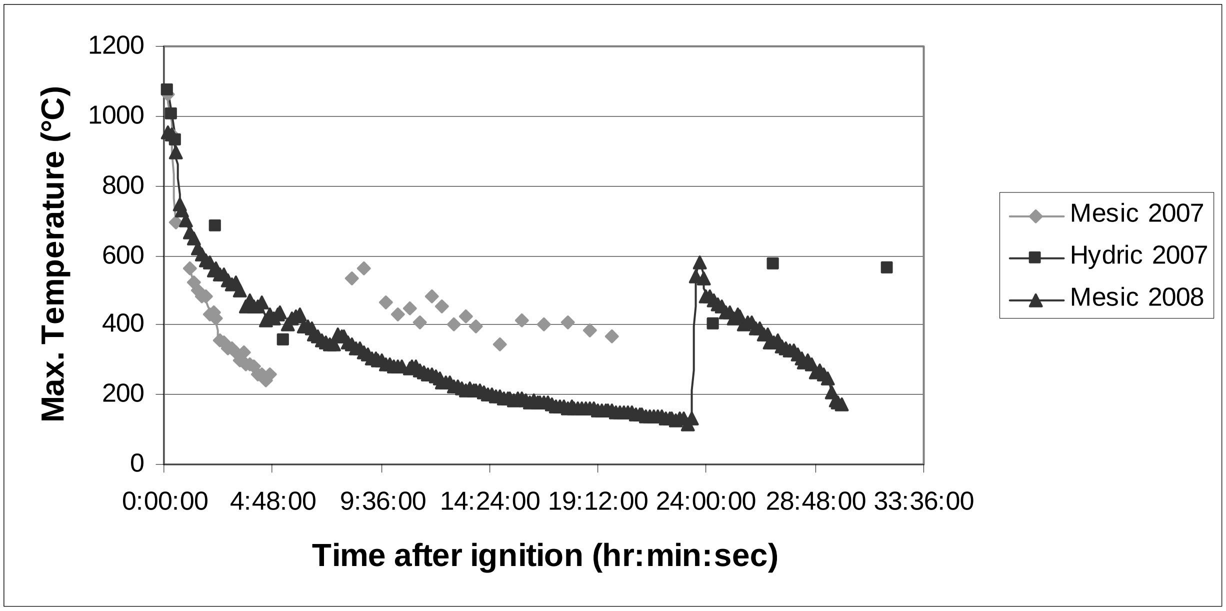


Figure 2.6 Maximum aboveground temperatures during three slash pile burns. Legend refers to soil moisture type and burn year. The thermal camera was removed during the 2007 burns to take images of other piles and repeatedly returned, whereas in 2008 the camera was focused on the pile for the length of burn. The second rise in temperature (seen at 24 hours for Mesic 2008) corresponds to when repiling was done to encourage more complete combustion.

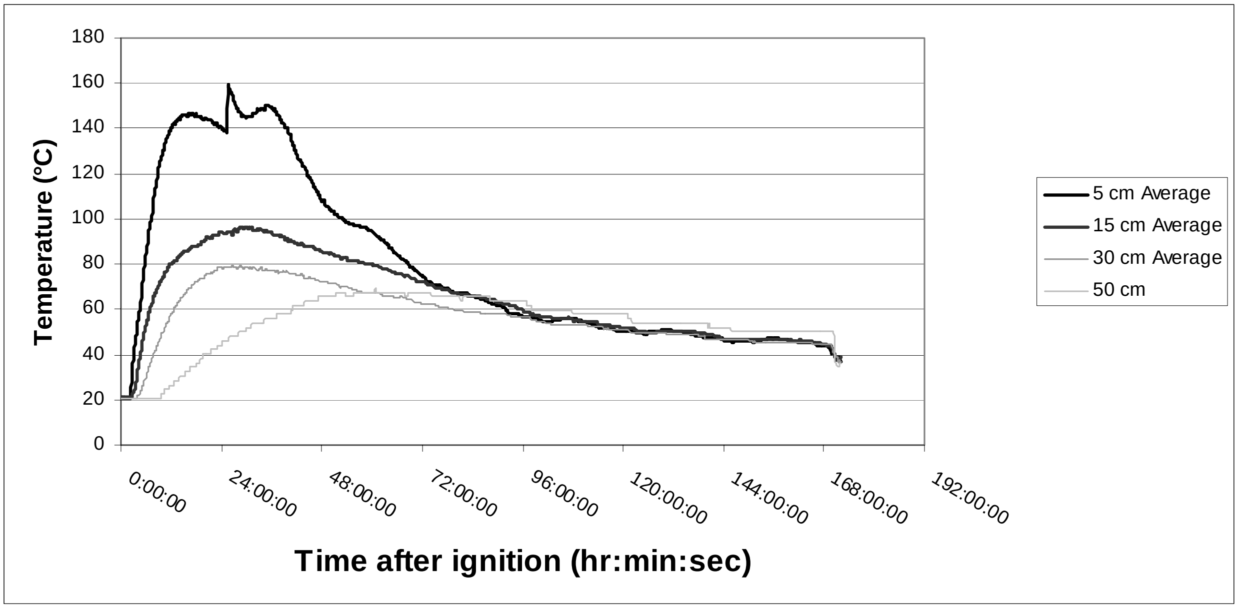


Figure 2.7 Example of a temperature-time profile by soil depth. Average belowground temperatures were obtained from data loggers situated along a 10 m transect below one pile burn.

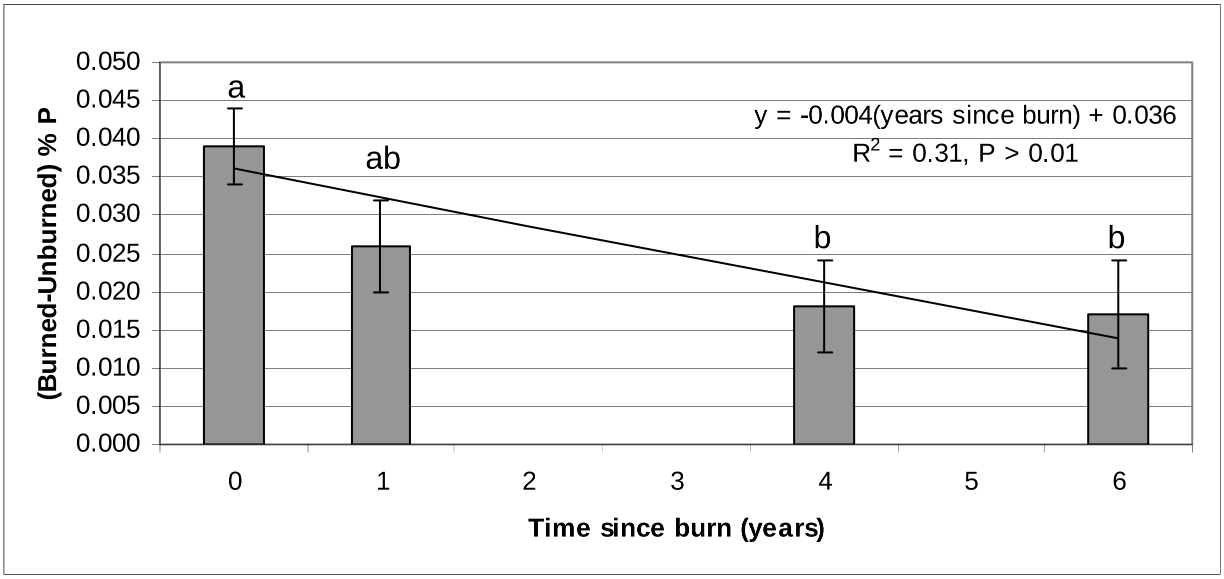


Figure 2.8 Change in difference in P (%) between burned and unburned conditions over time. Significant differences between burn years ($p<0.05$) determined by analysis of variance are labeled.

Figure 2.9 Change in difference in pH between burned and unburned conditions over time. Significant differences between burn years ($p<0.05$) determined by analysis of variance are labeled.

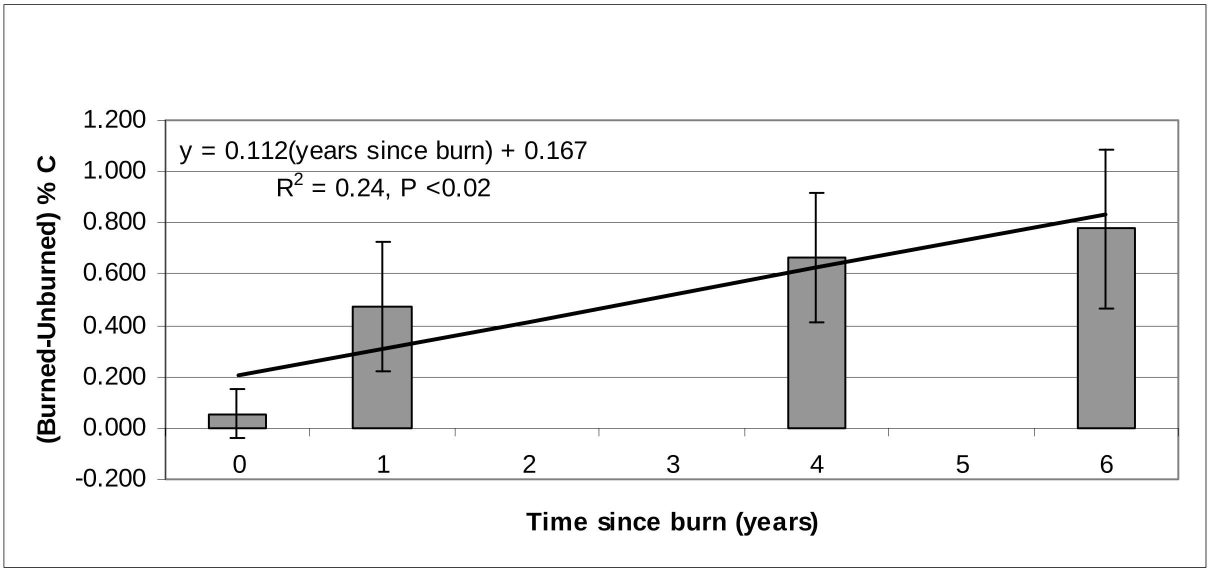


Figure 2.10 Change in difference in C (%) between burned and unburned conditions over time. No significant differences among burn years for difference terms were noted by analysis of variance.

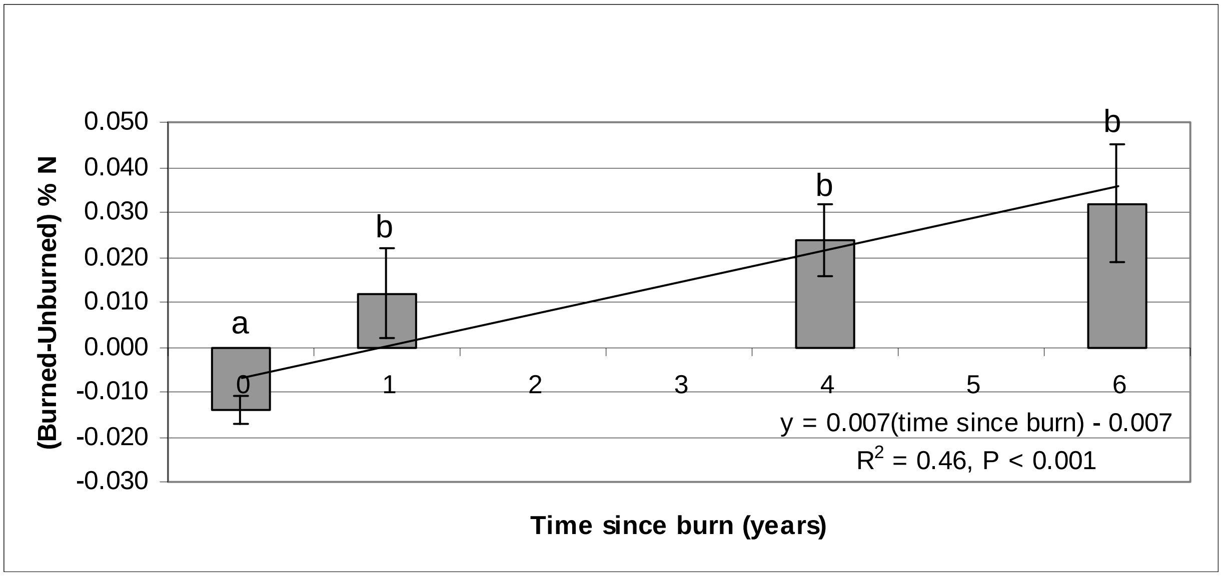


Figure 2.11 Change in difference in N (%) between burned and unburned conditions over time. Significant differences between burn years ($p<0.05$) determined by analysis of variance are labeled.

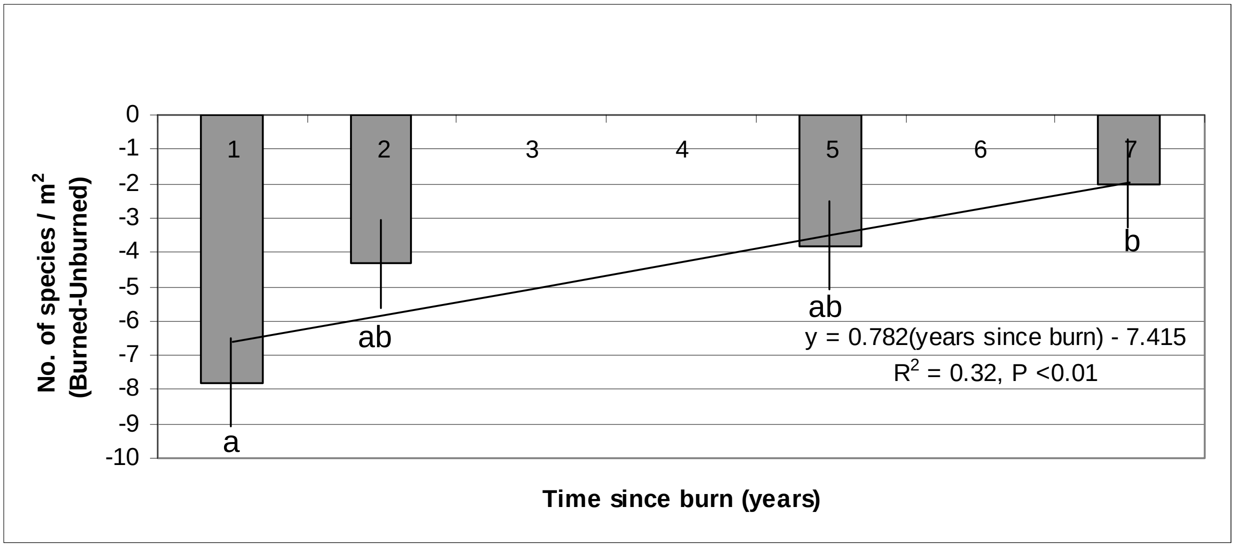


Figure 2.12 Change in difference in species richness between burned and unburned areas with time since burn. Significant differences between burn years ($p<0.05$) determined by analysis of variance are labeled.

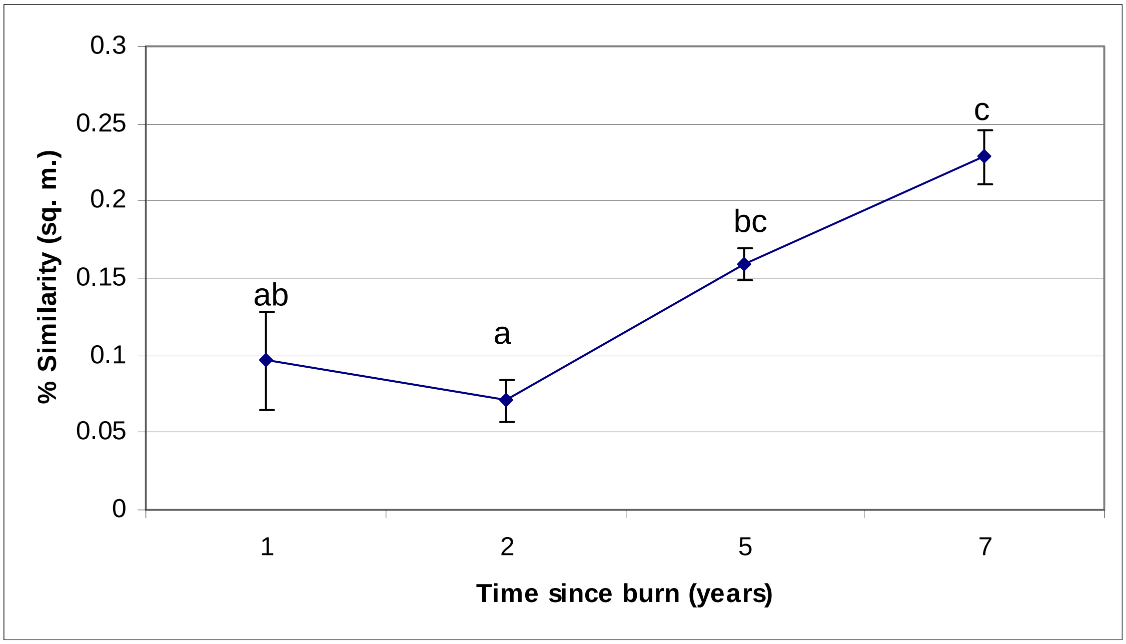


Figure 2.13 Vegetation percent similarity (Jaccard) between burned and unburned locations with time since burn. Regression performed on rank-transformed data was significant ($p<0.0001$) with time since burn. Significant differences between burn years ($p<0.05$) determined by analysis of variance are labeled.

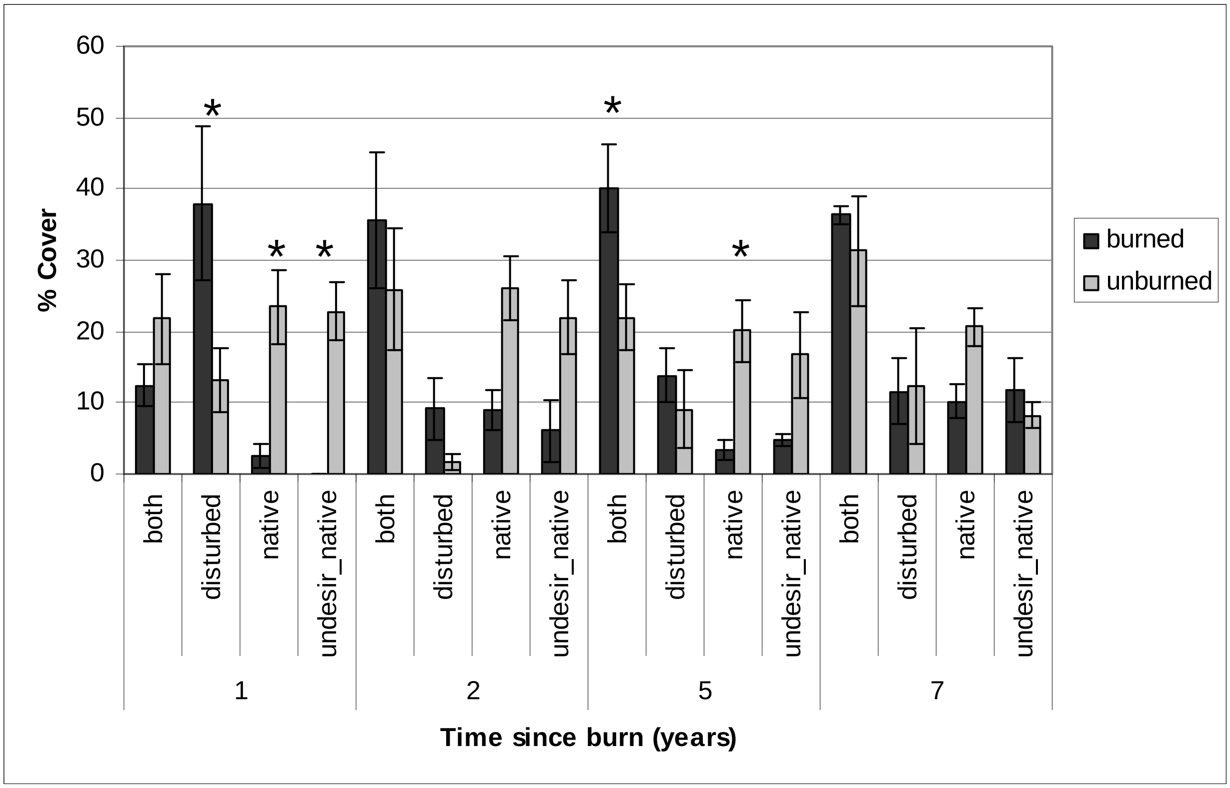
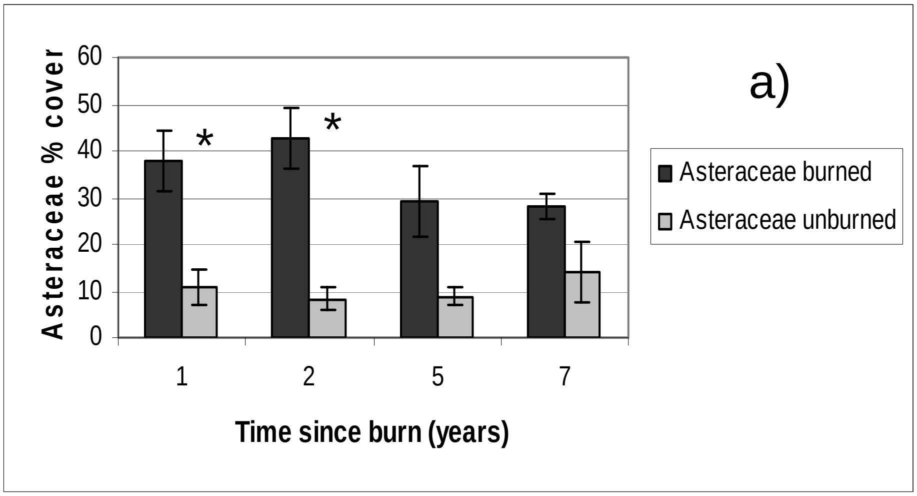


Figure 2.14 Percent cover of disturbance classification categories (modified from Wunderlin and Hansen 2003) showing burned and unburned values by time since burn. * indicates significant difference at $p<0.05$ between burned and unburned levels within a given year.



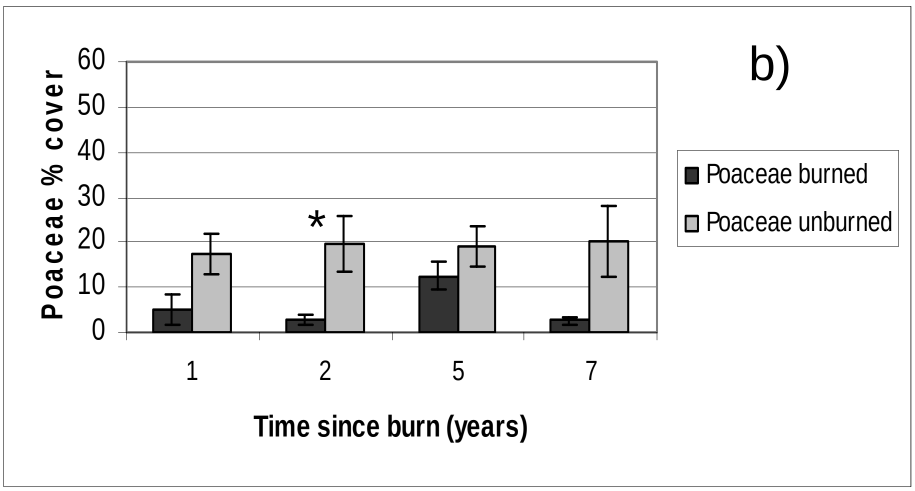


Figure 2.15 Percent cover of Asteraceae a) and Poaceae b) in burned and unburned areas with time since burn. * indicates significant difference ($p<0.05$) between burned and unburned levels for a given year.

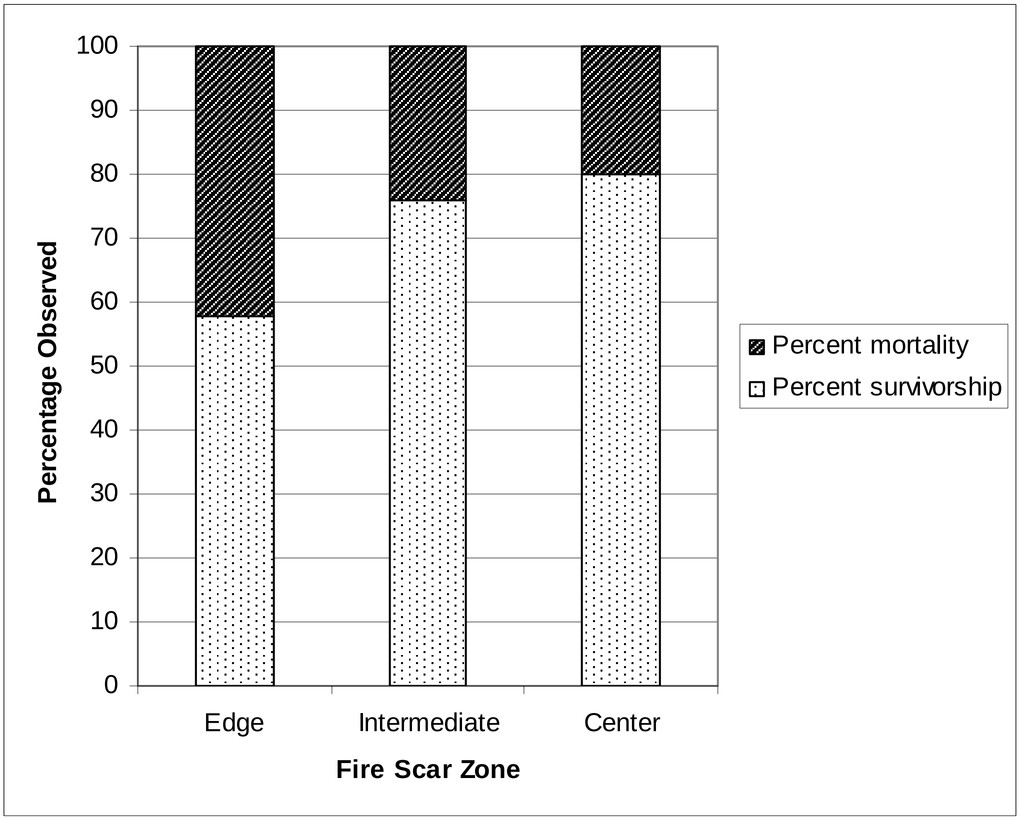


Figure 2.16 First-year survivorship of longleaf pine seedlings planted in fire scars, by zone.

CHAPTER 3

NATIVE REVEGETATION POTENTIAL OF SLASH PILE BURN SITES IN THE LONGLEAF PINE ECOSYSTEM[1]

[1] M. N. Creech. To be submitted to *Restoration Ecology*.

ABSTRACT

Restoration efforts that require tree thinning or removal to achieve a more open canopy or to remove undesirable species produce the need for residual slash debris maintenance to eliminate potential wildfire hazard. Slash pile burns may be a prominent feature in restoration of fire-maintained ecosystems affected by hardwood encroachment as a result of fire suppression, such as in the longleaf pine-wiregrass ecosystem. Yet the establishment of a ruderal plant composition within these fire scar sites may cause subsequent prescribed fire in the landscape to burn patchily, creating the potential for renewal of undesirable hardwood establishment. Thus, further ground cover restoration may be required to restore fire corridors. The objective of this study was to examine the potential for reintroduction of native ground cover species into fire scars. Specifically, we examined the following questions: i) Does survivorship of wiregrass vary across a gradient of soil conditions associated with fire severity within fire scar sites? ii) Is survivorship and growth of wiregrass regulated by an amendment of native topsoil to the fire scar? iii) Is seed germination inhibited by soil conditions associated with fire scars?

We found that survivorship of wiregrass in fire scars with or without native topsoil amendment was equivalent, but that greater biomass growth occurred in wiregrass plants which received native topsoil amendments, particularly in the areas of the fire scar most severely affected by fire. Germination and establishment of some native species introduced as seeds to fire scar soils was also increased by native topsoil amendment. Thus we conclude that planting seedlings or seeds with native soil amendments could provide opportunities to accelerate the re-establishment of native species that are important for the use of prescribed fire. Reconnection of the fire corridor across slash pile burn areas within the longleaf pine landscape could help advance the larger restoration objectives.

INTRODUCTION

Restoration efforts that require tree thinning or removal to achieve a more open canopy or to remove undesirable species produce the need for residual slash debris maintenance to eliminate potential wildfire hazard. An economical and commonly used silvicultural practice is to arrange the unmerchantable wood debris into large piles that are later burned. Fire scars from these intensive fires, however, cause near complete eradication of the soil seed bank and are typically recolonized by ruderal or non-native plant species, which then may compromise restoration objectives (Korb et al. 2004; Haskins and Gehring 2004; Chapter 2).

Slash pile burns may be a prominent feature in restoration of fire-maintained ecosystems affected by hardwood encroachment. In the case of the *Pinus palustris* Mill.—*Aristida stricta* Michx. (longleaf pine-wiregrass) system, slash pile burns may occur following removal of unmarketable hardwood species (e.g. *Quercus virginana* Mill. [live oak], *Quercus laurifolia* Michx. [laurel oak] and *Quercus nigra* L. [water oak]) that have gained dominance as a result of fire suppression, substantially altering forest structure and ground cover species composition (Ware et al. 1993; Kane et al. 2008). Without frequent fire, fire-intolerant hardwoods are able to grow large, produce seedlings able to proliferate in the absence of top-killing conditions from fire, and deposit leaf litter that suppresses fine fuels of the herbaceous ground cover (Waldrop et al. 1992; Ware et al. 1993). The increased canopy closure and leaf litter prevent further longleaf pine recruitment. Restoration of longleaf pine-wiregrass to fire-suppressed areas thus incorporates hardwood removal followed by renewal of frequent prescribed fire to prevent hardwood encroachment, as a means to restore structure and function to the system (Provencher et al. 2001; Brockway et al. 2005).

As a result of the potential high density of burn scars within a forest and the establishment of a ruderal plant composition within these sites that persists for many years (Chapter 2), subsequent prescribed fire may burn patchily. For example, in large tracts of second-growth longleaf pine savanna with intact native ground cover in southwest Georgia, fire scars were dominated by ruderal members of the Asteraceae family (Chapter 2), which do not provide sufficient fuels for fire compared to native grasses (Jack and Conner 1999; Mitchell et al. 2006). Thus, fire scars may provide locations in which hardwood re-establishment can proceed upon seed introduction, without inhibition by fire.

Delayed establishment of native ground cover species to the severely burned areas may be attributable to persistent altered soil characteristics such as initially sterilized soil conditions, elevated available nutrients and increased pH (DeBano et al. 1998; Certini 2005; Chapter 2). Nutrient levels, such as available N and P may also vary within the fire scar, being comparatively higher in center areas compared to edges (Korb et al. 2004; Jiménez Esquilín et al. 2007). With heavy ash deposition, salinity levels may be so high as to inhibit plant growth for 1-2 years after a burn (Menzies and Gillman 2003). Additionally, the microbiota that recolonize fire scars have a different composition than adjacent soils in the months to years after burn, and fungi are slower to return than bacteria (Renbušs et al. 1973; Jiménez Esquilín et al. 2007). Native plant species that form associations with soil microbes may find their common associates absent in fire scars, posing complications for native species reintroductions. Thus, native topsoil additions may facilitate native species return (Korb et al. 2004).

The dominant ground cover species in the longleaf pine ecosystem, *Aristida stricta* [Michx.] (wiregrass), is an important fine fuel but does not readily return to sites after it is removed by disturbance (Clewell 1989; Mulligan et al. 2002), and had not recolonized any fire scars in on-site evaluations (M. Creech, personal observation). Consequently, introducing wiregrass in the form of viable seed or seedlings is likely the only way to bring about the return of this functionally important species where it has been removed. Other native ground cover species may be limited by dispersal mechanisms from recolonizing disturbed sites as well (Hedman et al. 2000; Kirkman et al. 2004).

The objective of this study was to examine the potential for reintroduction of native ground cover species into fire scars. Specifically, we examined the following questions: i) Does survivorship of wiregrass vary across a gradient of soil conditions associated with fire severity within fire scar sites? ii) Is survivorship and growth of wiregrass regulated by an amendment of native topsoil to the fire scar? iii) Is seed germination inhibited by soil conditions associated with fire scars?

METHODS

Study Site

This study was performed within the 11,700 ha property of the J. W. Jones Ecological Research Center at Ichauway in southwestern Georgia (Baker County) (lat 31°15' N, long 84°30'W). This region within the Lower Coastal Plain is characterized by karst topography and elevations ranging between 30 to 100 m above sea level (Goebel et al. 2001). Daily temperatures in summer vary between 21-34°C, and in winter between 5-17°C, with a short winter and long growing season. Precipitation, which averages 131 cm annually, is evenly distributed throughout the year.

Ichauway includes 9,700 ha of second growth longleaf pine uplands having a diverse native ground cover including wiregrass, and 2,000 ha comprised of depressional wetlands, pine plantations, food plots and old fields. Much of the land was managed with frequent prescribed fire throughout the 20th century, and prescribed fire intervals currently average every 2 years. An extensive longleaf pine restoration project employing hardwood removal was begun in 2001 to reduce hardwood encroachment that had developed over previous decades (Stober 2008) and as of 2007 has been conducted over 3,600 ha.

Fire scars in this study were selected from both well-drained upland sites (loamy sands, classified as fine-loamy, kaolinitic, thermic Typic Kandiudults) as well as poorly-drained upland depressions (fine sandy loams, classified as fine, kaolinitic, thermic Typic Paleaquults) to compare revegetation outcomes between soil moisture regimes (Soil Survey Staff, NRCS 2009). All fire scars chosen were burned in July 2007, and revegetation treatments were applied in 2008. The fire scars had an average diameter of 20 m of bare soil. Wiregrass naturally occurred 10 to 50 m from all fire scars selected.

Aristida stricta Seedling Revegetation Study

We planted container-grown wiregrass seedlings (9 months old) into well-drained (n=6) and poorly-drained (n=4) fire scars in April 2008, in a split-plot design in which fire severity was the whole plot factor and topsoil treatments (with or without) were applied in each fire severity location. Each fire scar was stratified into three concentric zones according to varying levels of fire severity (center, intermediate, and edge). Center plots were established within 3 m of the center; intermediate plots were generally established between 3-7 m from the center, and edge plots were set up beginning 1 m inside the fire scar edge. We established paired topsoil treatment plots within each concentric zone of each fire scar, for a total of six plots per fire scar (2 treatments x 3 zones) (Figure 3.1). In each plot (0.8 m^2), we planted 30 seedlings (in 6 rows with

5 seedlings across, and 20 cm spacing between seedlings). A total of 180 seedlings were planted in each fire scar among the possible zone*treatment combinations. Seedlings were grown from native seed harvested on Ichauway.

To implement the soil amendment treatment with topsoil, native topsoil was collected just before planting from natural areas having wiregrass that were in close proximity to a given fire scar. Enough soil in the 0-20 cm depth was removed from numerous points within an approximately 9 m^2 area to fill a 5-gallon bucket, and homogenized in a larger bin, for application at each fire scar. Some wiregrass roots and individuals of smaller-sized species were removed during the process, but less than 10% of the removal area was affected. Topsoil was added to “with topsoil” treatments at the base of the dibbled planting hole prior to placing in a wiregrass seedling, as well as spread over the top of the wiregrass root bundle upon seedling introduction. The microbial composition of native topsoil was not identified. For “without topsoil” treatments, no native topsoil was added with planting.

Following planting, water was supplied twice weekly when necessary, in the absence of precipitation, from mid-April to mid-June 2008 in all plots (two gallons of water applied per plot). The study area was in a period of drought at the time of planting, and received 46.7 mm of precipitation from 15 April to 15 June 2008, compared to an average of 196.4 mm over the same time period in previous years (1971-2000) (University of Georgia Automated Environmental Monitoring Network 2009). Pine straw was spread below plants in all treatment plots in mid-June 2008 to help retain soil moisture.

Early survivorship of wiregrass seedlings was assessed in late June 2008. After 90% survival was found for both treatments (with or without topsoil), any dead wiregrass seedlings were replaced with live ones in late June 2008. Two fire scars of the poorly-drained soil type experienced prolonged flooding in August 2008 following a tropical storm, causing mortality of wiregrass seedlings in these fire scars, and they were removed from analysis.

Survivorship of seedlings was again assessed in October 2008. Flowering status of all seedlings was noted at this time. An individual was counted as flowering if at least one tiller flowered. Aboveground biomass of all wiregrass seedlings was collected. The plant material was oven-dried at 70°C for 48 hours, and dry weights determined.

Seed Mix Study

Native seed germination and establishment in burned and unburned soils was examined in a shadehouse environment. Soil samples were removed from 12 fire scars (six from each soil type) in July 2008 at 1 year post-burn. At each fire scar, one sample was removed from each of the three zones (center, intermediate, and edge), and one unburned sample was taken 5 m from the pile edge, to serve as an unburned soil control. A second form of control using potting soil was included. The fire scar zones and controls together will be referred to as separate "locations". Individual pots were labeled as to their respective location and their soil type (well-drained or poorly-drained) and, according to these two factors, were considered replicates as part of a randomized incomplete block design (incomplete because we blocked by locations but not by all piles). Two topsoil treatments (with or without topsoil) were assigned randomly but equally among the five locations and two soil types, with three samples associated with each location*soil type combination, for a total of 60 samples.

For each soil sample, an intact volume of soil was extracted, 35 cm long x 29 cm wide by 10 cm deep to fit the dimensions of sample containers. We placed the soil core directly into the container with as little disturbance as possible. All containers were sown with a seed mix of 6 native species. Although both the topsoil applied as well as the unburned control soils contained seed from a soil seed bank, the species used in the soil mix were not present in a corresponding seedbank study (Chapter 2). For the topsoil addition treatment, we spread a 0.95 L volume of homogenized topsoil over the surface of the extracted soil sample, creating a 1-2 cm deep layer, prior to adding the seed mix. Seed mix packets contained 180 seeds, represented by 6 native species in the following percentages of the total seed mix: [Grasses: *Aristida stricta* (25%), *Sorghastrum secundum* (25%); Legumes: *Lespedeza angustifolia* (20%), *Desmodium floridanum* (10%), *Tephrosia virginiana* (10%); and Composite: *Rudbeckia hirta* (10%)]. Legume seeds of *L. angustifolia* and *T. virginiana* were recently scarified to break the hard seed coat. Seed was pressed lightly into soil. Sample bins were watered daily for 3 months. At the end of the study in October 2008, we counted the number of seedlings established for each species in all sample bins.

Analyses

Wiregrass seedling survivorship, flowering status, and biomass data were analyzed separately using mixed models (PROC MIXED, SAS Institute Inc., version 9.1), according to a split-plot design to test for differences among fire scar zones and soil amendment treatments. Where interaction terms were significant, contrast analyses were performed in PROC MIXED. Flowering status was evaluated as a possible covariate with biomass. In the seed mix study, seedling establishment counts by species were compared by soil type, topsoil treatment and location, using three-way analysis of variance (PROC GLM, SAS Institute Inc., version 9.1). Where heterogeneity of variance occurred in seed mix data (for *S. secundum* percent establishment), data were rank-transformed prior to non-parametric analysis.

RESULTS

Aristida stricta Seedling Revegetation Study

No differences in wiregrass survivorship or growth were found between soil types (well-drained or poorly-drained) ($F= 1.70$, $p<0.19$) and, thus, data were combined for further analyses. Survivorship averaged 91 ± 0.04 % SE, and did not differ by treatment or zone (Figure 3.2).

Increasing wiregrass biomass was evident along the burn gradient towards center zones with topsoil additions (Figure 3.3). Fire scar zone, topsoil treatment, and the interaction term of zone*treatment were all significant for determining wiregrass biomass (Table 3.1). However, we performed tests of effects slices which demonstrated that biomass differed between treatments at center zones but did not differ at edge zones (Table 3.2). Moreover, mean biomass of topsoil-treated wiregrass was found to increase from edge to center zones, but the no topsoil-treated wiregrass biomass did not differ among zones (Table 3.3).

Topsoil treatment had a strong influence on flowering response ($F=9.82$, $p<0.01$). The application of native topsoil resulted in mean percent flowering 1.6 times greater than the without topsoil treatment (Figure 3.4). However, neither fire scar zone nor zone*topsoil treatment ($F=1.45$ $p>0.20$; $F=0.89$ $p>0.40$) were factors affecting flowering. Mean percent flowering per plot was positively related to total biomass per plot ($F=28.52$, $p<0.0001$).

Seed Mix Study

Seedling recruitment responses to soil type, location (fire scar zone or control), and topsoil treatment were species-specific. Soil type affected establishment of only two of the species, *S. secundum*, ($F=9.37$, $p<0.01$), and *D. floridanum* ($F=5.76$, $p<0.05$), with higher rates of both species occurring on mesic soils. Neither species was responsive to other factors (p>0.15). For *L. angustifolia*, the effect of topsoil addition, and an interaction term of topsoil treatment and location, were significant (treatment: $F=4.39$, $p<0.05$; location*treatment: $F=3.13$, $p<0.05$). Seedling recruitment was markedly reduced in soils of center and intermediate zones in the absence of topsoil, but with topsoil addition, levels of establishment in center and intermediate locations increased to become equivalent to the higher levels found in edge and unburned locations. We found significant contrasts ($F=4.00$, $p<0.001$) to support the role of topsoil additions as increasing establishment in center and intermediate areas for this species, but not in edge or control areas where establishment was initially higher. Recruitment of *R. hirta* increased in response to application of topsoil regardless of zone ($F=20.63$, $p<0.0001$), while *T. virginiana* had reduced establishment in center and intermediate zones which was not improved by topsoil addition ($F=4.66$, $p<0.01$). No wiregrass germinated in any treatments.

DISCUSSION

Aristida stricta Seedling Revegetation Study

The high percent survivorship of *A. stricta* seedlings, regardless of their location along the burn gradient or type of soil treatment (with or without native topsoil), indicates that wiregrass can be successfully reintroduced into slash pile burn sites and suggests an opportunity to hasten the re-establishment of fine fuels necessary to carry prescribed fire.

The fact that wiregrass seedling growth without topsoil addition did not change across the fire scar, in spite of a significant nutrient gradient, implies that factors other than available N and basic cations were limiting. Phosphorus is not likely to have been limiting given the five-fold increase of total P after burn; however, given the basic conditions that resulted from the burn, some level of P sorption to Ca may have occurred in the ash layer (Kwari and Batey 1991; Fisher and Binkley 2000). Even so, P sorption is unlikely to have occurred along the entire seedling root zone, because of more neutral conditions ($7 < pH < 8$) below the ash layer (Chapter 2). The interacting effect of topsoil and burn gradient on biomass and flowering of wiregrass indicates that increased nutrient uptake was promoted only under certain conditions. The causal mechanism for increased biomass of wiregrass with topsoil is unclear, but some attributes of topsoil are likely more important than others. Specifically, the presence of topsoil would introduce low levels of nutrients, presumably reduce soil pH around the wiregrass root zone, and supply microbial contributions to fire scar areas (Moorman and Reeves 1979; Holmes 2001; Korb et al. 2004).

Arguably, a small amount of nitrogen addition with topsoil, coupled with a surface temperature gradient from non-vegetated soil in the center to the more vegetated edge of the burn scar, could have led to increased mineralization of topsoil N in close proximity to wiregrass roots (Ojima et al. 1994; Knoepp and Swank 2002). Wiregrass, a C_4 plant, has particularly high nutrient use efficiency (Sage 2004; West and Donovan 2004). However, the N contribution in topsoil is not the most likely reason for the progressive increase in growth and flowering where topsoil was added based on the fact that the available N present in the fire scar from 0-2 years after burn was probably much higher at all locations than the amount mineralized from topsoil (Covington et al. 1991; DeBano et al. 1998; Korb et al. 2004; Jiménez Esquilín et al. 2007). Consequently, we contend that nitrogen inputs from introduced topsoil would be minimally important.

One alternative explanation is that topsoil additions could reduce the elevated soil pH around the seedling root zone and potentially alter nutrient availability, enabling wiregrass to better respond to components of the fertility gradient from edge to center. However, the most plausible explanation for the nutrient uptake response may be the introduction of microbial associates missing from the pile burned area. Evidence supporting the role of microbial associates is provided from other studies of fire scars. In a study of microbial response along a pile burn gradient, Jiménez Esquilín et al. (2007) reported that fungal biovolumes in pile burn centers were nearly non-existent at 15 months after burn and remained reduced at the edges. At the same time, bacterial biovolumes were reduced, though less drastically, in pile centers compared to pile edges (Jiménez Esquilín et al. 2007). Furthermore, native topsoil introductions in pile burned soils have been shown to result in higher potential arbuscular mycorrhizal fungi root colonization levels in bioassay plants grown in those soils, than in areas not receiving topsoil (Korb et al. 2004). This finding implies that although fungi are naturally slow to return to pre-burn levels or composition, some fungi can persist in pile burn soils if introduced.

Arbuscular mycorrhizal (AM) inocula have been noted in soils of longleaf systems, as a uniform presence of AM spores (Henkel and Kindell 1991), and in terms of moderate levels of mycorrhizal inoculum potential in soils (D. Sylvia and D. Gordon, personal communication) based on *Zea mays* bioassays (Sylvia 1994). Although AM root colonization in wiregrass specifically has been documented in some environments of the longleaf pine ecosystem (Kindell and Alden 1993, unpublished data; Jenkins 2003), it has been found lacking in others (Anderson and Menges 1997), and thus the significance of these associations for wiregrass growth and survival is unknown.

The presence of increasing amounts of char toward the center of the pile may also facilitate microbes, including those introduced with topsoil, given char's high water holding capacity (Tryon 1948; Pietikäinen et al. 2000) and porosity which can act as a refuge for bacteria and fungal hyphae (Zackrisson et al. 1996). In one study, microbial biomass was found to be greater in humus when it was combined with charcoal than when it was alone (Zackrisson et al. 1996), and the strong nutrient adsorption properties of char were found to be partially attributable to the high microbial colonization it experiences, such that microbes actively decompose adsorbed compounds on char surfaces and create more space for char to adsorb further substances (Zackrisson et al. 1996). Although nutrient limitation can occur at high applications of char, char can improve long-term soil fertility in general, with slow release of adsorbed nutrients (Lehmann and Rondon 2006).

Although the mechanism of initiation of flowering in wiregrass is also not well understood, reproduction of wiregrass is strongly associated with timing of fire, with increased rates of flowering and seed production when burned in the growing season (Clewell 1989; Streng et al. 1993; Mulligan et al. 2002). Thus, flowering in the absence of a growing season burn is surprising, but may be related to a first-year flowering phenomenon in wiregrass plants commonly observed in container-grown seedlings immediately after outplanting (L. K. Kirkman, personal observation). The positive relationship between biomass and percent flowering suggests that factors that encourage growth, such as nutrient availability, may have also encouraged flowering. Increased levels of K have been found important in wiregrass seed development (Kalmbacher 1983; Pfaff et al. 2006), and available P may contribute to reproduction, as has been noted in other species (Marschner 1995); both of these nutrients are likely to be present in hardwood ash.

The high rates of survivorship of seedlings regardless of treatment may be partly explained by the fact that they were grown in non-sterile potting soil and were therefore somewhat buffered from the altered soil conditions in fire scars. Regardless, the high survivorship demonstrates that wiregrass is capable of establishment and growth in fire scars immediately after burn. Wiregrass is a very hardy and long-lived species (Clewell 1989; Mulligan et al. 2002), and these findings suggest that wiregrass is tolerant of a wide range of soil characteristics, from the weakly acidic soils of its natural range, to a more nutrient-rich and basic fire scar environment.

Seed Mix Study

Topsoil additions enhanced germination of some species (*L. angustifolia* and *R. hirta*) and had no effect on the others. Species that responded to topsoil may have been sensitive to the presence of ash, which has been found to inhibit seed germination in other species, particularly in the first years after burning. For example, ash cover (3-5 cm deep), and heat exposure in combination with ash cover, reduced the emergence of seedlings from soil seed banks in Mediterranean pine forest, while heat exposure alone facilitated germination (Izhaki et al. 2000). Similarly, ash produced from pile burning has been shown to reduce germination and growth of some crop species for two seasons after burn (Menzies and Gillman 2003; Pitman 2006). Korb et al. (2004) found seed additions combined with topsoil to lead to twice the native plant cover compared to seed only treatments which were introduced directly to ash, suggesting that ash and/or a sterilized soil environment reduced the success of seed germination or establishment, while topsoil promoted it.

Our seed mix study was set up in a shadehouse environment, where water was not limited, and PAR (photosynthetically active radiation) was reduced relative to forest stand conditions. Thus, outcomes would likely differ somewhat with seed mix application in a pile burn scar environment, particularly in regard to potential water limitations.

Species-specific differences in response to the burn gradient and to topsoil in our study suggest that some species would benefit from topsoil additions. Furthermore, our results demonstrated that some native legumes common to the longleaf pine ecosystem are able to persist in recent pile burn scars if introduced. Successful establishment of legumes has been found in bio-char conditions because of their ability to compensate for lack of available soil N with biological N-fixation (Lehmann and Rondon 2006). Furthermore, N fixation rates may be enhanced under these circumstances due to elevated levels of P (Leidi and Rodríguez-Navarro 2000; Peet 2006). Available soil N is likely to decrease over time, while P is more likely to remain in soil indefinitely (DeBano et al. 1998). The lack of germination by wiregrass in the shadehouse is not clear and further trials in field conditions are recommended.

CONCLUSIONS

Wiregrass survivorship does not vary as a function of soil conditions associated with fire severity, but differences in growth were evident with native topsoil additions. Greater growth of wiregrass resulted from native topsoil additions in the most severely affected areas by burn. Seed germination for some species sown into fire scar soils was enhanced by topsoil additions.

The findings of this study, coupled with evidence from our studies of the long-term persistence of soil alterations in fire scars, provide strong evidence that reintroduction of seedlings or seeds with native soil amendments could provide restoration management opportunities to accelerate the re-establishment of native species important for the use of prescribed fire. Inclusion of native soil amendments, which may lead to enhanced growth, may also promote more continuous prescribed fire movement through fire scars and discourage hardwood establishment. Ultimately, reconnection of the fire corridor across slash pile burn scars within the longleaf pine landscape would advance the larger restoration objectives in this fire-maintained ecosystem.

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Table 3.1 Mixed-models analysis of variance for *A. stricta* biomass by fire scar zone and topsoil treatment.

Effect	DF	Den DF	F Value	Pr>F
Zone	2	14	6.48	0.0102
Treatment	1	21	8.25	0.0091
Zone*Treatment	2	21	3.78	0.0395

Table 3.2 Effects slices from mixed-models analysis of variance for *A. stricta* biomass by fire scar zone and topsoil treatment.

Effect	Zone	Treatment	Num DF	Den DF	F Value	Pr>F
Zone*Treatment	Center		1	21	14.89	0.0009
Zone*Treatment	Edge		1	21	0.03	0.8688
Zone*Treatment	Intermediate		1	21	0.9	0.3529
Zone*Treatment		Top-	2	21	0.64	0.5364
Zone*Treatment		Top+	2	21	10.04	0.0009

Table 3.3 Separate paired contrasts of treatments "with topsoil" (Top+) and "without topsoil" (Top-), and zones (center, intermediate, and edge), for *A. stricta* biomass.

Contrasts	Num DF	Den DF	F Value	Pr>F
Contrast 1				
Top+ center vs. Top+ edge	1	21	19.63	0.0002
Top+ center and Top+ edge vs. Top+ intermediate	1	21	0.45	0.511
Contrast 2				
Top+ center vs. Top+ intermediate	1	21	7.81	0.0109
Top+ edge vs. Top+ intermediate	1	21	2.68	0.1167
Contrast 3				
Top- center vs. Top- edge	1	21	1.07	0.3123
Top- center and Top- edge vs. Top- intermediate	1	21	0.21	0.6502
Contrast 4				
Top- center vs. Top- intermediate	1	21	0.01	0.9062
Top- edge vs. Top- intermediate	1	21	0.84	0.37

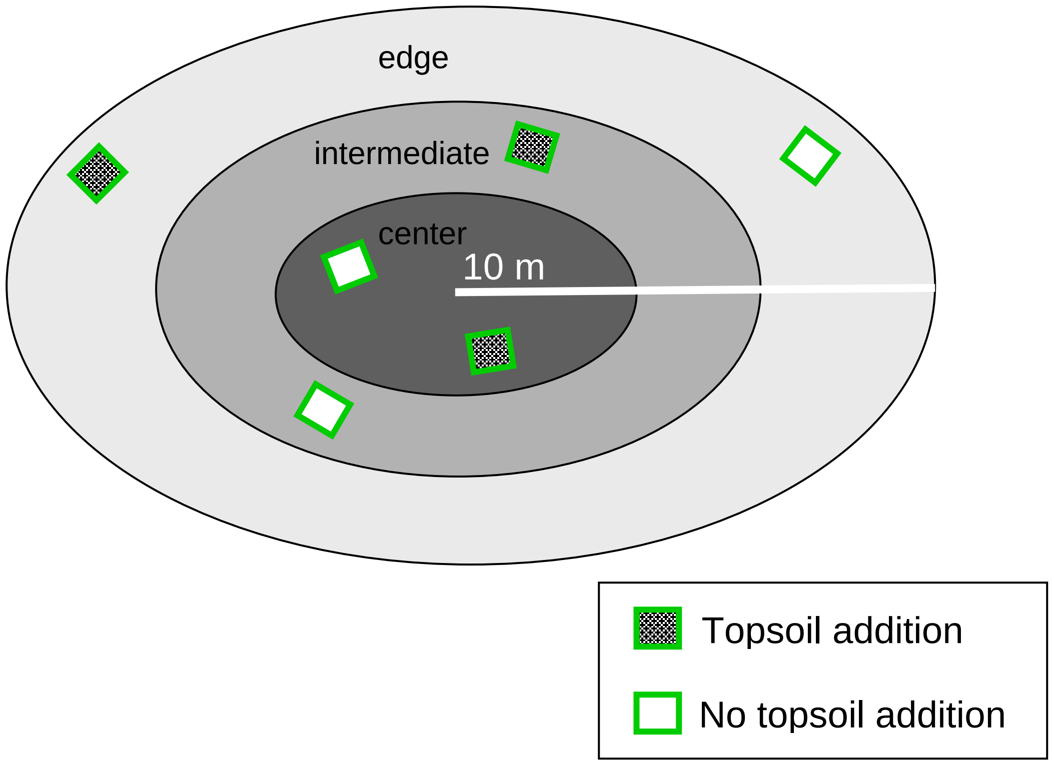

Figure 3.1 Wiregrass seedling planting design in concentric fire scar zones (edge, intermediate, and center) with or without native topsoil additions. Average radius of fire scars was 10 m from edge boundary to center point.

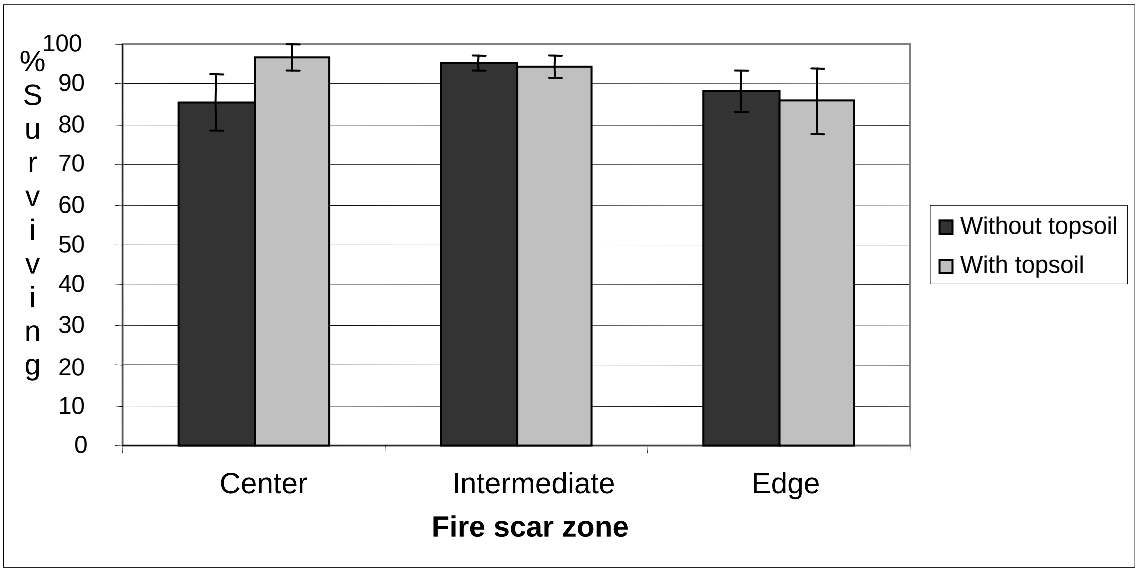

Figure 3.2 Percent survivorship of *A. stricta* by fire scar zone and topsoil treatment. There are no significant differences between zones or treatments.

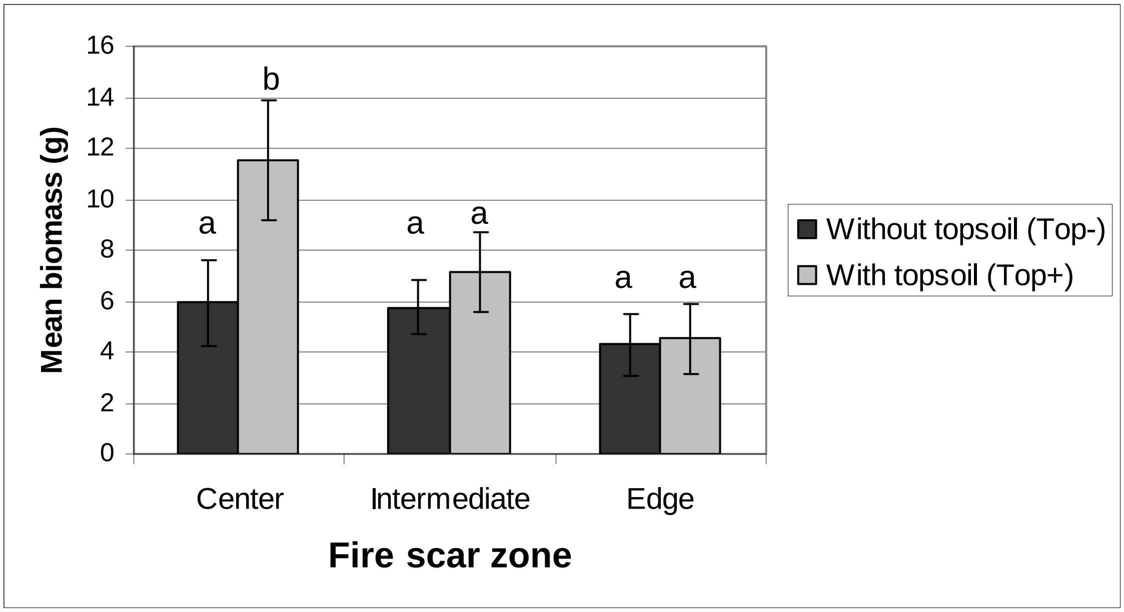

Figure 3.3 Mean *A. stricta* biomass by fire scar zone and topsoil treatment. Significant differences ($p<0.05$) are labeled.

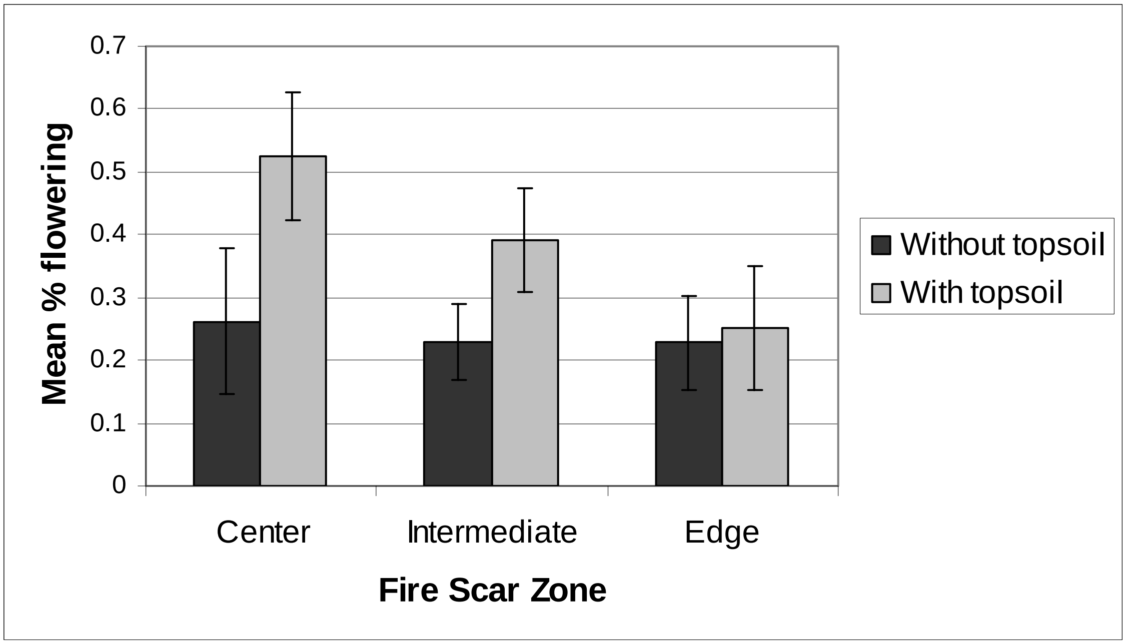

Figure 3.4 Mean percent flowering of *A. stricta* in fire scar zones by topsoil treatment. "With topsoil" treatment is significant (p=0.05). The interaction term of zone*treatment is not significant (p=0.42).

CHAPTER 4

CONCLUSIONS

The longleaf pine ecosystem is of high priority conservation interest in the southeastern U.S., having been reduced to less than 5% of its original extent (Frost 1993; Outcalt 1996; Mitchell et al. 2006). In our study area, hardwood cover increased over the previous decades on both field edges and in interior woodlands (Goolsby 2005), and removal of hardwoods has been employed in both types of areas, followed by the use of slash pile burns for disposal of wood debris, on over 3,600 ha. of the property.

We demonstrated that slash pile burns cause persistent changes in the soil environment, including increased phosphorus and pH, as well as nitrogen and carbon, which may allow recolonizing vegetation to have greater biomass growth and reproduction than in adjacent less nutrient-rich unburned areas. Seed bank sterilization occurs as a consequence of the high temperatures produced in these burns, such that re-vegetation cannot arise from a persistent, native seed bank. Recolonizers are typified by native, wind-dispersed, ruderal species. Few native species distinctly associated with ground cover of undisturbed longleaf pine stands returned to fire scars, even after 6 years. In particular, the reduced dominance of grass cover in fire scars within a large-scale longleaf pine restoration context is important relative to the presence of fine fuels and the prospect of reintroducing prescribed fire management in these areas, and in long-term control of hardwood encroachment.

To restore structure and function of the longleaf pine ecosystem, an open longleaf pine canopy and ground cover are required that allow for frequent prescribed fire to prevent hardwood succession (Palik and Engstrom 1999; Brockway and Outcalt 2005; Kirkman et al. 2007), fed from both longleaf pine needle cast and native herbaceous species such as wiregrass. Slash pile burn scars lack both of these fuel types to carry fire. Thus, the absence of fire to control hardwood seedling establishment, even in localized areas, has consequences for the larger system (Jacqmain et al. 1999; Goolsby 2005).

Hardwood succession is the natural result in this system without the frequent use of prescribed fire (Ware et al. 1993), and there are many pathways that result in hardwood establishment (Jacqmain et al. 1999; Goolsby 2005), which is likely a consequence of the inherently heterogeneous nature of fire (its path determined by variable fuels and weather) together with the small window of time that hardwood seedlings will succumb to fire before they become fire-resistant (e.g. 2-4 years; Jacqmain et al. 1999). These factors are amplified by the extensive fragmentation (roads, wildlife food plots, fields) in this system that compartmentalizes prescribed fire efforts with uneven edge effects of burning, where hardwoods may become established. Also, traditional use of annual cool-season burns may be less effective in reducing hardwood abundance, and frequent burning with reduced fuel loads may contribute to burn patchiness and safe sites for hardwood establishment, particularly in wetter sites (Waldrop et al. 1992; Jacqmain et al. 1999; Goolsby 2005). The use of slash pile burns within longleaf restoration areas creates scarred locations that in future years are likely to have insufficient fuels to achieve the prescribed fires necessary to perpetuate dominance of the longleaf pine system (Figure 4.1).

We demonstrated that, although important native species such as longleaf pine and wiregrass do not typically return to fire scars independently within the time-frame needed to promote prescribed fire, they are capable of surviving in the altered soil conditions. The results of the wiregrass and longleaf pine seedling planting and seed mix studies further reveal that native topsoil amendments added to the center and intermediate areas of fire scars where alteration is greatest facilitate native vegetation establishment and growth. Rapid establishment of fine fuel vegetation in the year after planting would encourage more uniform prescribed fire over the area (Figure 4.2). Based on our results, we recommend that reintroduction of seed or seedlings and amendments of native topsoil be considered as a potential restoration tool.

We did not demonstrate the mechanism for enhanced growth in wiregrass with topsoil application, an area which warrants further study given the importance of this ground cover species within the longleaf system. The microbial environment in fire scars is likely to offer an explanation, coupled with the altered soil nutrient characteristics, and the extensive deposition of char. Furthermore, examination of the adsorption capacity of char and the associated microbial community may help identify important factors in the long-term retention of nitrogen in the soil system.

Slash pile burns have been a feature of forestry management in the southeastern U.S. for a long time and are likely to continue in some form in the future, and this study provides management suggestions where longleaf pine restoration is desired. However, other environmental concerns associated with the use of slash pile burns related to air quality standards and emissions of CO_2 may become more critical in the future. Thus, economically feasible alternatives for the removal of wood debris, as a wood product or biomass use for energy, are likely to become necessary.

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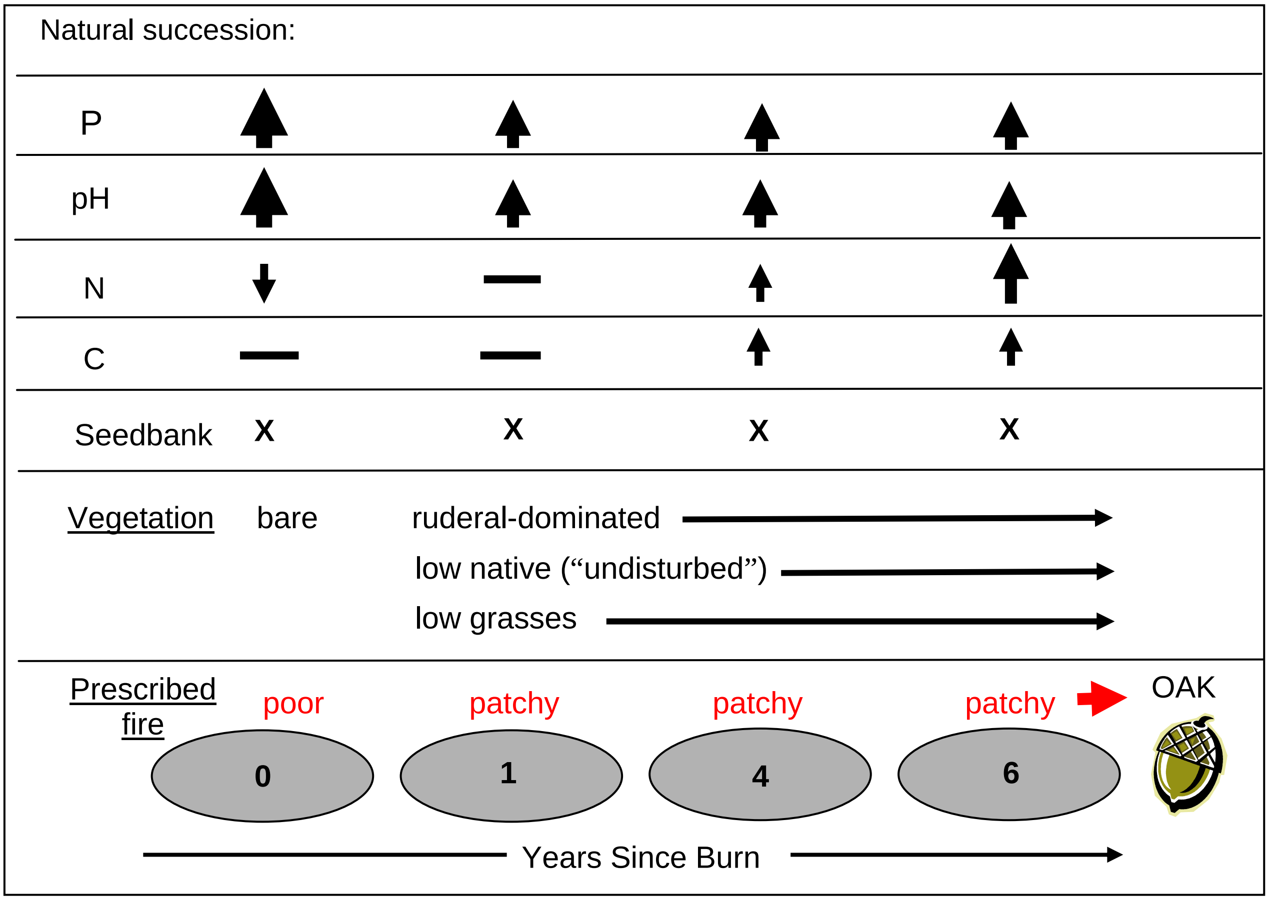

Figure 4.1 Model of slash pile burn characteristics over time, with potential for prescribed fire given natural vegetation recolonization.

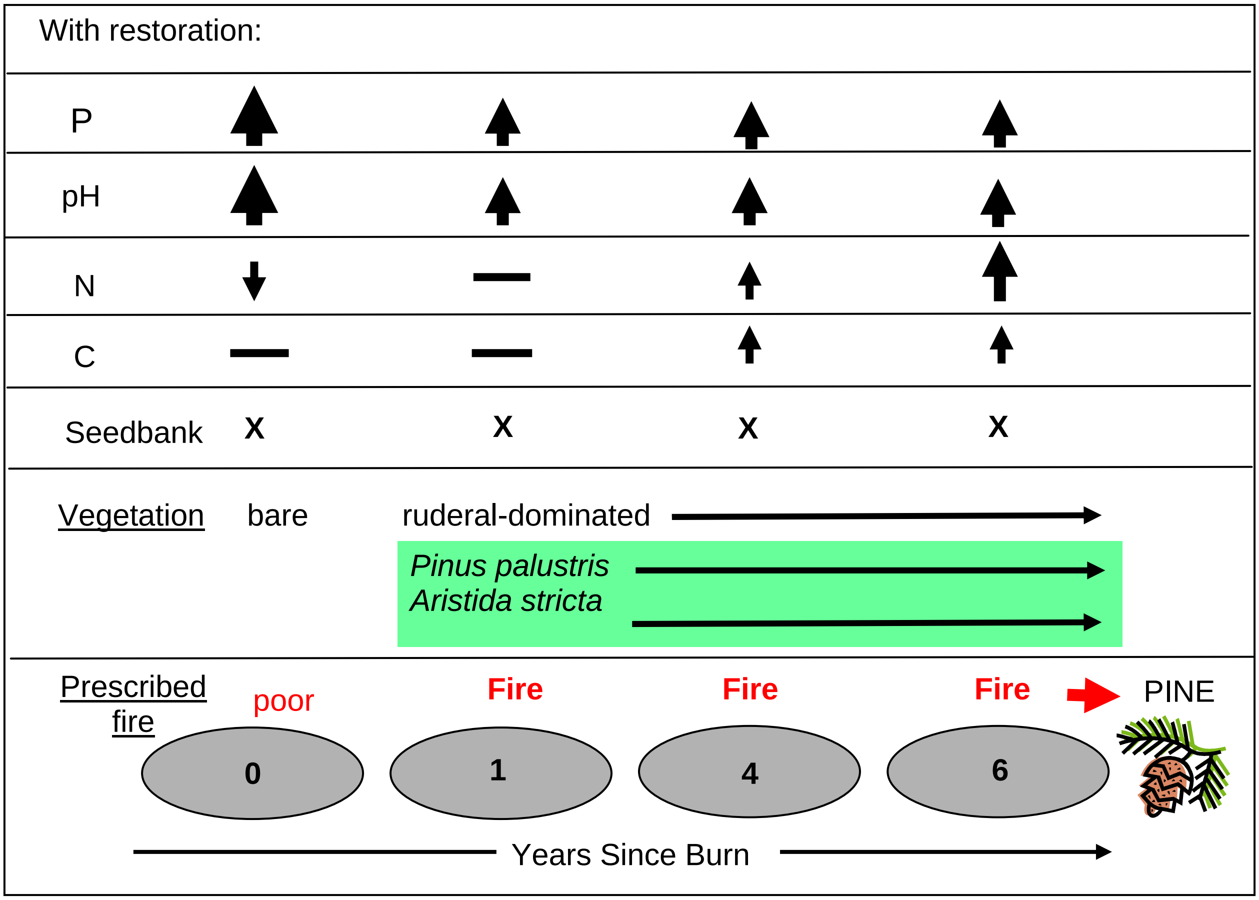

Figure 4.2 Model of slash pile burn characteristics over time, with potential for prescribed fire given the reintroduction of longleaf pines and wiregrass.