Fuel management practices and their effects on forest and grassland soils of the eastern US.

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1. Introduction

Fuel management treatments in the eastern US encompass a diverse array of activities which have a range of potential impacts on the soils within watersheds of managed forests and grasslands. In industrial or production forests the predominant fuel management strategies are intensive site preparation (bedding, roller chopping, burning slash), use of herbicides, and pre-commercial or early rotation thinning, and these activities probably impact the most land area in the east. On public lands managed for natural resources, the fuel treatment strategies often are more varied and can include the use of herbicides and thinning operations, use of prescribed fire, grazing, or targeted chainsaw-felling of specific understory species. Thus, effects of fuel management on forest soils can be very subtle or protracted, such as a plant-soil-microbe feedback resulting from removal of a single plant species, or effects can be acute and profound such as the direct soil-profile disrupting disturbances associated with site prep and logging operations. Because the functions of forest soil arise through complex interactions between its physical, chemical and biological components, we will address effects of individual fuel treatment practices on each of these soil properties.

There is a wide range of different ecosystem types occupying the eastern North America, and this diversity is reflected in the underlying soils. Eastern soils differ from one another across broad ranges of climatic conditions, parent material, topography (elevation and aspect), age, disturbance history, and the biota that they support – all factors which influence the long term development of soil, and which ultimately determine the type of soil found in a given location (Jenny, 1941). Soils in the Eastern US fall into nearly every soil order, and are classified into hundreds of soil series (Chaper X, this volume, Clinton et al.). Here we attempt to review the effects of fuel management practices (specifically prescribed fire and mechanical fuel treatments) on soils of eastern North America, and we include data from as many different ecosystem types and soil types as possible. The reviewed material is therefore necessarily very broad in scope, and for the purposes of this review we attempted to collect and synthesize the available soil-related data from as many different ecosystem types from the eastern United States as possible.

2. Prescribed Fire Effects on Eastern Soils

Prescribed fire is probably the most widely used treatment for fuel reduction in the eastern United States. These fires may be applied to logging slash as a component of site preparation for new plantings, or they may be applied as surface fires to reduce understory vegetation, or to promote certain desirable plant and animal species. Furthermore, fire serves a crucial functional role in many (if not most) wildland ecosystems of the eastern US. This relationship is particularly well known in the pine-dominated ecosystems of the Atlantic and southeastern Coastal Plains (230 – Subtropical Ecological Division), and equally so in tallgrass prairie ecosystems in the Midwest (250 – Prairie Ecological Division) . Prescribed fires are also increasingly used for fuels management in the pine forests of the Lake States (210 – Warm Continental; 220 – Hot Continental), but less is known about their effects on ecosystem properties. Finally, although the role of fire in eastern hardwood forests (primarily in 220 – Hot Continental and M220 – Hot Continental Mountains) is less well known than for pine forests, much work has been performed in recent years to shed light on this important question.

2.1 Physical effects of prescribed fire (Summarized in Table 1)

The predominant physical effects of fire on forest soils include heat transfer, development of hydrophobic conditions, increased soil temperature, enhanced risk of erosion, and degradation of soil aggregate structure. Heat transfer and hydrophobicity in soils are closely linked because heat causes volatilization of waxes and oils in organic material which diffuse into soils and then condense around soil particles causing them to be water repellent. The degree to which this process occurs is dependent upon fire temperature, residence time, and the nature of the organic matter in the forest floor (DeBano 2000). In general, the development of hydrophobicity in soils of eastern North America is not a serious problem in relation to prescribed fires, and we were unable to find any documented cases of this phenomenon in the east.

The degradation of soil aggregate structure as a potential physical effect of prescribed fires has been hypothesized for Oak savanna ecosystems of Missouri, but this phenomenon has yet to be directly measured (Rhoades et al. 2004). These authors suggested that destruction of aggregate structure might partially explain observations of slow recovery of plant communities in soils where logs had 'burned out' in a prescribed fire. Such aggregate destruction may be related to the changes in soil texture observed in their study, as well as changes in water infiltration and water holding capacity of soils impacted by intense "burn-outs" of large downed fuels. In any case, the net watershed effect of such impacts will be dependent upon the amount of large down wood in burned areas and how these materials are consumed in prescribed fire events.

Increased soil erosion has been observed in wildfire impacted areas, but evidence for large soil losses due to erosion in areas subjected to prescribed fires is limited. For example, in relatively steep slopes (35-45%) in the southern Appalachian mountains (Hot Continental Mountains – M220), Swift et al. (1993) observed localized movements of soil in a treatment area that had been burned in a prescribed fire, but there was no net soil loss from the treatment area. These authors attributed the sediment retention observed in their study to entrapment of sediments in the remaining intact forest floor which was only 66% (or less) consumed in the fires. Perhaps more important than erosion of soil from the landscape is erosion associated with fire control activities, and in particular the use of plowed fire lines in the application of prescribed fire management (Van Lear et al 1985).

2.2 Chemical effects of prescribed fire (Summarized in Table 2)

2.2.1 Carbon

Pools of carbon likely to be affected by prescribed fire include plant roots, total soil organic C, microbial biomass C, and "black" C (charcoal and soot). All of these pools of C are more or less tightly related to one another, and changes in one pool in response to fire will be expected to be associated with changes in one or more other pools. The magnitude of fire effects on soil C pools is largely dependent upon the intensity and frequency of fires, soil type, and forest type.

2.2.1.1 Plant root C

A large proportion of management-induced changes in soil organic matter carbon can be traced to cumulative effects on C dynamics associated with plant roots. Among other management practices, prescribed fire can strongly influence the plant community found in forested stands, and this depends largely on the frequency and intensity of fire. In general terms, the shorter the fire return interval, the more prevalent perennial grasses become in understory vegetation. This pattern is typical of mesic grassland systems, for example in tallgrass prairies of eastern Kansas (Prairie Division – 250), where there is a clear relationship between fire frequency and the cover of warm-season perennial grasses (Knapp et al. 1998), and a corresponding increase in the amount of total root biomass in frequently burned soils compared to unburned soils (Kitchen et al. 2009). Because grass root tissues typically have very wide C:N ratios, the decomposition of this material is slower than analogous root tissue from forbs or woody species, with the net effect of greater accumulations of total soil organic C in systems with greater relative cover of warm season grasses (Knapp et al. 1998).

Increases in perennial grass cover with frequent fire are also well known from forested systems such as longleaf pine systems (Subtropical Division – 230) on the Atlantic and Gulf Coastal Plains (Glitzenstein et al. 2003; Brockway and Lewis 1997), loblolly and shortleaf systems on the Southern Piedmont (Phillips and Waldrop 2008), and shortleaf/bluestem systems in the Ouachita Mountains of Arkansas (Subtropical Mountains – M230) (Liechty et al. 2005). In other systems where fire return interval is longer, or where fire has been excluded for a long period and prescribed fires have only recently been reintroduced, there has been little documented change in understory plant community with fire. This has been the case in hardwood forests in Ohio (Hot Continental – 220) (Hutchinson et al. 2005), and in jack pine systems in Ontario (similar to those found in the Great Lakes states, Warm Continental Division – 210) where prescribed (site preparation) fires reduced grass cover in the first year following fire, but effects were negligible two years after fire (Tellier et al. 1995).

2.2.1.2 Soil Organic C

One of the long-term consequences of increased inputs of grass derived detritus is the accumulation of soil organic C. This is particularly true for grassland soils which have long been noted for their high organic matter content, but this pattern holds for any system with extended periods of increased grass cover. This organic matter accumulation in soils with a large component of grass in the understory is the result of the much higher C:N of grass material. The C:N of organic matter is of critical importance because high C:N material takes longer to decompose, and gives rise to more recalcitrant forms of OM in the later stages of decomposition (with potential to ultimately change the amount of C stored in a particular soil profile). Thus, the net effect of frequent prescribed fire is increased inputs of organic matter which often have longer turnover time (relative to OM in unburned systems), and thus, an indirect effect of prescribed fire is an increase in the net storage of C in mineral soil horizons. Other forms of soil organic C that are influenced by the occurrence of prescribed fire include microbial biomass C and charcoal and soot (black carbon or BC), and these pools are discussed below.

2.2.1.3 Black C

Not all ecosystem carbon subjected to combustion in prescribed fire is volatilized to CO_2. Depending on the fire severity, a fraction of the carbon will remain in the ecosystem in the form of highly recalcitrant carbon (i.e. black carbon). The importance of black carbon (BC) in the total C cycle of fire-impacted ecosystems is increasingly being recognized (DeLuca and Aplet 2008). However, several aspects of the input and cycling of BC, for example in response to different fire frequencies, have not been thoroughly examined. Charcoal, elemental C, and soot derived from biomass burning are generally considered as a recalcitrant pool with a very long turnover time from centuries to millennia (Deluca and Aplet 2008). The chemical interactions between BC and other organic matter constituents (microbial pools, humus, soil organic matter, and fresh litter), however, are complex and not well studied (with a few notable exceptions such as Wardle et al. 2008; and the review by Czimczik and Masiello 2007). Available published data for BC formation and its interactions are primarily derived from ecosystems with long fire return intervals (e.g. DeLuca and Aplet 2008; Wardle et al 2008), and these systems likely will have BC dynamics very different from the pine savanna systems of the southeastern US. We have observed formation and storage of BC in the mineral soil horizons of a longleaf pine flatwoods site with an annual fire regime (Callaham et al., unpublished data), and this is expected to significantly affect the net storage and turnover of C in these systems.

2.2.2 Nitrogen

Nitrogen (N) is frequently the limiting nutrient in forested ecosystems, and this element occurs in many different forms that can be influenced by fire. Nitrogen is an integral part of all biomass in ecosystems, and N concentrations in organic detritus (or necromass) is highly influential on the rate of decomposition of this material (Coleman et al. 2004). Finally, the inorganic forms of N (nitrate, NO_3^-; and ammonium NH_4^+), and the rate at which these forms are released from detritus or supplied by N-fixing plants and microbes usually has a profound influence on the overall fertility of a given soil volume.

Prescribed fire can have dramatic effects on N cycling, particularly when fires are frequent. One of the principal effects that prescribed fires have is the volatilization and loss of N from the organic horizons of soil. This effect is directly related to the intensity of the fire and the relative proportion of the Organic-horizon (O) that is consumed in combustion. Also important is the temperature at which combustion occurs, and the depth to which high temperatures penetrate the organic horizons. For example, in a laboratory study, Gray and Dighton (2006) found that the temperature at which different litter materials were burned had strong influence on the amount of N volatilized. Temperatures above 400°C resulted in 90-100% loss of N whereas temperatures between 100-200°C retained at least 75% of the original N content. The long-term consequences of the loss of N can be significant, either through chronic loss from frequent repeated fires, or from a large loss in a single high-severity fire. For example, Elliott et al. (2002) demonstrated that at a site which had experienced a more severe site preparation fire (with relatively large proportions of organic horizons consumed), had lower tree seedling growth several years after the fires, than did sites with less severe fires, and this effect was attributed to the loss of N capital from the system via volatilization.

In other aspects of N supply and cycling, however, prescribed fire has been demonstrated in many systems to have a positive effect. For example, N mineralization (microbial processing of organic N into plant available mineral forms) is either not affected by prescribed fire or is increased following prescribed fires (as seen in Table 2). The net overall effect of prescribed fire on N dynamics in soil is most likely a function of fire frequency and intensity. Very frequent, or very intense fires are likely to have negative effects on total N, but fires of intermediate frequency or lower intensity may produce positive effects in terms of N availability.

2.2.3 Phosphorus (Summarized in Table 3)

Phosphorus (P) is often the second most limiting nutrient in forested ecosystems, and its availability is also influenced by prescribed fire. As a major component of ash, it should not be surprising that P would be affected by occurrence of fire, but the chemistry of P in soils is highly complex and usually is strongly influenced by the pH of soil. Because P is chemically bound to aluminum and iron oxides at low pH, and similarly, is bound to calcium at higher soil pH (Schlesinger, 1991), the availability of P in ash is somewhat dependent upon the pH of the underlying soil. Further complicating the chemistry of P in relation to fire is the fact that the ash produced by the fire has other constituents that themselves can change the pH of soil, at least in the short term. Thus, depending on the pH of soil before and after fire, the availability of P will be variably affected. In general, for pine dominated soils (and indeed for most forest soils in eastern North America), the pH is typically in the range where P becomes chemically bound with Iron (Fe) and Aluminum (Al) (i.e. pH 5.7 and below), and the tendency for ash addition would be to temporarily increase the soil pH to a more favorable condition relative to P availability. However, such effects are usually short-term (on the order of months to a couple of years) as the capacity of soil to buffer changes in pH is very large. Finally, it is notable that at very high temperatures (> ~770°C), P can be volatilized and lost from ecosystems (Neary et al. 1999), and as such, fire intensity can be of great importance to overall P availability following prescribed fire.

2.2.4 Other Cations (Summarized in Table 3)

In addition to the two macronutrients already discussed (N and P), there are several other essential nutrients that may be affected by the incidence of prescribed fire in forested landscapes. The most widely studied of these are cations such as calcium (Ca), magnesium (Mg) , potassium (K) and sodium (Na). All these cations serve critical functions in various aspects of plant cell metabolism, and thus their availability for uptake has the potential to influence site productivity and even plant community composition to some extent. Because cations are typically not subject to volatilization in fires, their availability generally goes up after a fire, when ash is deposited into the soil. Again, because biological demand for these cations is relatively high (i.e. plants, microbes and animals all compete for these nutrients), the duration of fire-mediated spikes in availability is typically short and on the order of weeks to months.

2.3 Biological effects of prescribed fire (Summarized in Table 4)

2.3.1 Plant roots and fire

There is a large amount of information available that details the effects of fire on plants in eastern forests . Effects range widely, from completely positive to completely negative effects of fire on plants. This range of effects depends largely on the community of plants present in a forested landscape (fire tolerant species, fire sensitive species, or a mixture), and on the intensity of the fire (low intensity prescribed fire, high intensity wildfire, or something in between). Fire almost always results in the death of some plants in a given system, and the extent to which plants are killed has a strong relationship to the effects of fire on roots. The killing of fire sensitive plants aboveground results in an input of dead roots belowground, and this input of new material has the potential to influence the decomposers (microbes) as well as the entire soil food-web at least in the short term.

Another effect of prescribed fire on plant roots is that there is a change in root distribution throughout the soil profile. In grasslands such as tallgrass prairie, annual fire has the effect of causing roots to be distributed more deeply throughout the soil profile (Kitchen et al. *in press*). In forested ecosystems, data on fire effects on root distribution is scarce, but evidence from longleaf pine systems suggests that frequent prescribed fire has similar effects on fine root distribution in mineral soil. In longleaf pine systems where fire is excluded in the long-term, fine roots proliferate in the organic horizons of the soil, but in frequently burned sites, the O-horizons are much reduced or eliminated completely, and thus fine root biomass is increased in mineral soil horizons (O'Brien et al, *in review*). The degree to which prescribed fire affects root distribution in other eastern ecosystems has not been extensively studied.

2.3.2 Soil microbes and fire

The effects of fire on soil microbes in eastern forests seems to be dependent to a large extent upon the intensity of the fire. The responses of soil microbes to fires range from no detectable effect in the case of low intensity prescribed fires to total sterilization of the surface layers of soil in very hot wildfires (see Joergensen and Hodges, 1970; and Renbuss et al., 1973). This early work focused primarily on the abundance of microorganisms and not their activity levels. However, workers have observed that although there may be a decrease in abundance of microbes following fire, the remaining microbes can have activity levels that are greater than that of the microbial community prior to the fire (Poth et al. 1995). These authors, working in tropical savanna systems in Brazil, found that the increased rates of microbial processes, such as denitrification and production of methane and carbon dioxide, persisted for one year following fire, but the nature and duration of microbial responses to fires in eastern forests are not well known. In one study examining soil CO_2 efflux (the combined production of CO_2 from plant root respiration and microbial and soil animal respiration) in loblolly pine stands of the South Carolina Piedmont, Callaham et al. (2004) observed that soil respiration (one indicator of microbial activity) was decreased in plots that had been burned or thinned and burned, and these authors attributed this result to warmer soils in these two treatments, along with increased inputs of belowground detritus in the form of dead plant roots.

Most of the more recent work on soil microbes and their responses to fire has made use of new techniques designed to facilitate examination of the diversity or functional capacity of the microbial community. The most frequently used approaches are enzyme-based assays of microbial activity, and the community carbon utilization profile (CCUP). Enzyme assays utilize the actual concentrations of ecologically important enzymes in soils to make inferences about the make-up and function of the microbial community at the time of sampling. CCUP uses an array of different carbon sources to evaluate the potential metabolic capacity of the microbial community from the sampled soils. These carbon utilization profiles give an estimate of how diverse the microbial community functions are – if the microbes from a site can use more of the different carbon sources in the assay, then that community is considered functionally more diverse. In general, changes in the concentrations of enzymes in soil can be attributed to changes in the relative importance of various functional groups of microbes. There are a large number of such enzymes present in soil but a few are particularly well characterized and have standardized methods of measurement (Tabatabai 1982). In reference to fire effects on soils several enzymes have been used as response variables, and these include acid phosphatase (indicative of total microbial biomass, and phosphorus mineralizing organisms), phenol oxidase (indicative of white rot fungal biomass), chitinase (indicative of bacterial decomposition of more recalcitrant organic matter), aryl-sulfatase (indicative of microbes processing sulfer containing organic matter), a-glucosidase (indicative of fungal metabolism of cellulose and hemicellulose), and L-glutaminase (indicative of microbes involved in metabolism that results in nitrogen mineralization). Results from enzyme assays in studies comparing burned to unburned soils seem to indicate shifts in the microbial community towards a community that is geared toward metabolizing more recalcitrant materials, but these results are somewhat site dependent and responses differ in terms of duration after fire (see summary in Table 4).

2.3.3 Microinvertebrates and fire

In one of the few studies dealing with microinvertebrate responses to fire in eastern forests, Metz and Farrier (1971) reported a general reduction of microarthropods (mainly Collembola and mites) with increasing prescribed fire frequency in loblolly pine stands on the coastal plain of South Carolina (Division 230 – Subtropical). In this study, the authors compared the abundance of microarthropods in plots that had either been burned every year, burned every 3 to 4 years, or left unburned for many years. The main results from this study were that abundances of mites and springtails were reduced by a small amount (~25%) by periodic prescribed fires, but that this reduction was dramatic (75-80% fewer) when prescribed fires were conducted annually. Similar studies in midwestern forests (Division 220 – Hot Continental) have shown similar results in that reduction of litter mass with prescribed fire generally results in reductions of microarthropod numbers (Dress and Boerner, 2004; Brand 2002). The consequences of these reductions for the decomposition of new leaf litter have not been thoroughly addressed.

The response of microarthropods to fire has also been studied in many other systems including eastern grasslands such as the tallgrass prairie systems in eastern Kansas and Oklahoma (Division 250 – Prairie). These studies have generally found that microarthropods are decreased in abundance with frequent fire (Seasteadt et al. 1984). This negative effect of fire is mostly attributed to decreased habitat for mites and springtails, because many of these organisms live in decomposing leaf litter, and much of this litter is lost in fires.

2.3.4 Macroinvertebrates and fire

There have been few scientific studies of the responses of soil invertebrates to fire in forested ecosystems of the eastern US. Of the few studies that have addressed these organisms' responses, the general pattern observed is that the response is often driven by changes in habitat structure, or by changes in the amount or the quality of food resources. Thus, whenever fire affects vegetation, temperature or moisture, or the nutrient status of a soil, there is potential for impact on the soil invertebrate community. These impacts are not always predictable, as demonstrated by a study of ground and litter dwelling arthropods conducted by Hanula and Wade (2003). These authors found that the frequency of prescribed fires (plots burned annually, every two years, every four years, or unburned for 40 years) in longleaf pine flatwoods of northern Florida had dramatic effects on numerous organisms. Interestingly, most of the arthropod groups collected during the five year study had negative responses to fire, but other groups were strongly favored by fire. For example, among 28 different spider groups that were collected, there were only four that responded positively to the frequent fires employed in the study.

Another study of litter dwelling and soil dwelling macroinvertebrates showed that the density of macroinvertebrates was significantly reduced one year after a prescribed fire in the upland forests of the Cumberland Plateau in Kentucky (Kalisz and Powell, 2000). Reduction in the number of beetle larvae accounted for a large proportion of the difference following fire, and the authors proposed that repeated fire in a single location could potentially have long-term negative effects on beetle populations and on the functions these beetles perform within the system.

Several studies have been conducted in grassland soils in eastern Kansas (Division 250 – Prairie) that focused on the responses of soil macroinvertebrates to fire. Studies have repeatedly shown that earthworms are strongly affected by fire in tallgrass prairie soils, and the usual pattern observed is for fire to increase the abundance of earthworms (e.g., James, 1982). Interestingly, in areas close to human habitations (i.e. with non-native earthworms present), prescribed fire has the effect of limiting the colonization of non-native earthworms into soils under frequently burned vegetation (Callaham et al. 2003). Results of this study suggested that the native earthworms in grassland soils are adapted to the warmer soil conditions frequently found under frequently burned vegetation, and that because fire improves the performance of grasses, the native earthworms may have strong habitat preferences for soils with abundant grass roots. This effect of fire on non-native earthworms may have potential application as a control strategy in eastern forests where invasions of European or Asian earthworms are currently underway, but this idea is in need of further research.

3. Mechanical Fuel Treatment Effects on Eastern Soils

3.1 Introduction

Mechanical fuels treatments (hereafter referred to as MFTs) have the potential to alter soil properties and processes dramatically, but under many conditions may have little to no impact on soils. Mechanical treatments affect soils through the use of heavy equipment, which may change physical and hydrological processes, and through the cutting and removal of vegetation and site organic matter, i.e, fuels, which changes soil fertility and soil chemical and biological processes (Powers et al. 1990). MFTs can vary from single-entry understory mowing or mulching treatments using small tractors, to multiple-entry whole-tree thinning and harvesting followed by harvest residue raking and piling (Table 5). In addition, MFT can occur under stand and soil conditions that are both resistant and resilient to impact or in conditions that provide little resistance to soil disturbance or nutrient removals and provide few mechanisms for recovery. No MFT is without the potential for impacting soil function, but conditions do exist under which any MFT can be used effectively without causing detrimental impacts to essential soil functions such as supplying adequate rooting medium, water and nutrient supply to plants, and water infiltration (without excessive runoff or erosion), .

In intensive production forestry, soil quality is restored or even improved following soil disturbance using other practices, such as soil tillage and fertilization (Fox 2000). These practices are feasible because they ameliorate damages and usually increase production. In extensive forest management systems, especially those in which timber yield is not the primary goal, the focus for managers is to minimize negative disturbances impacting soil productivity and rely on natural recovery processes and inherent site productivity (Grigal 2000). Therefore, a complete understanding of how MFTs affect soil properties and processes is necessary to avoid decreasing soil quality to the extent that natural processes cannot restore it.

Much of the basic knowledge we have regarding MFTs and soil impacts was developed quite some time ago, and most of the important foundational principles that describe how mechanical treatments impact soil were developed in agricultural and forestry systems. The crux of the ongoing problem is that the potential set of conditions to which the principles apply is virtually infinite, and it is only through continued, site-specific research that we will be able to better understand how to minimize negative impacts when mechanical forest management treatments are used. Therefore, we will only briefly review the basic concepts and widely accepted principles of soil disturbance effects and concentrate on describing the most current evidence available from studies from eastern U.S. forests.

3.2 Mechanical fuels treatments and soil physical properties and processes

MFTs have the potential to cause changes to soil physical properties and processes (see reviews by Lull 1959, Greacen and Sands 1980, Miwa et al. 2004), and these soil physical changes have been linked to reductions in germination (Pomeroy 1949), establishment and early survival of seedlings (Foil and Ralston 1967, Hatchell et al. 1970, Lockaby and Vidrine 1984, Bates et al. 1993, Scheerer et al. 1994, Brais 2001), sprouting or suckering success (Stone and Elioff 2000, Smidt and Blinn 2002, Stone 2002, Zenner et al. 2007), seedling root growth (Mitchell et al. 1982, Simmons and Ezell 1982, Tworkoski 1983, Simmons and Pope 1985, Jordan et al. 2003, Siegel-Issem et al. 2005), seedling shoot growth (Hatchell 1981, Lockaby and Vidrine 1984, Farrish et al. 1995), and growth of remaining trees (Moehring and Rawls 1970). However, soil disturbance and damage during mechanical operations is not a given (King and Haines 1979), and many studies have shown that soil physical disturbances do not necessarily lead to reduced tree survival or growth (Tiarks 1990, Reisinger et al. 1993, Xu et al. 2000, Carter et al. 2002, Sanchez et al. 2006, Scott et al. 2007). While the overwhelming majority of research on soil physical disturbance in eastern forests has been conducted in the South in pine forests or in the aspen forests of the north-central U.S., the general relationships hold for most forest types. Unfortunately, often general relationships are not useful in determining the impact across different site types or for particular soil functions within a given site type.

Several classification systems have been created to define soil disturbances. Most of these systems describe various degrees of harvest and/or forest floor removal and mineral soil disturbance. These classification systems have evolved from those defined by Dyrness (1965) for Pacific Northwest forests. Miller and Sirois (1986) and Aust et al. (1998) developed classification systems in the South, while Steber et al. (2007) recently used a nationally based system to evaluate disturbance in the Great Lakes region. These disturbance classification systems are used widely because unlike chemical or biological changes, soil physical disturbances have a clear and usually negative visual impact and because visual classification schemes provide an easy and rapid assessment of forest sites. Although visually based classification systems are useful to rapidly assess and monitor impacted areas, they are not generally effective at discerning quantitative changes in soil properties or processes (Aust et al. 1998, Steber et al. 2007). However, these systems are quite useful in determining the spatial extent of disturbance which is an important component to determining actual site disturbance.

Soil physical disturbances have generally been classified as compaction, rutting, and puddling or churning. Compaction occurs whenever the load applied to a soil is greater than its strength, and it results in an increase in bulk density (D_b) and a reduction in porosity. Mechanical traffic causes compaction when the soil contains enough water to reduce friction between soil particles and thus reduce soil strength, but not enough water to cause soil flow. Puddling occurs when the soil is wet enough to flow, traffic causes rutting, and repeated tire slippage smears pores and destroys soil structure (Miwa et al., 2004).

Bulk density is the most common method of quantitatively describing disturbance. Other properties and processes commonly affected by soil physical disturbance include soil strength (resistance to penetration, e.g., by roots), porosity and the distribution of pore sizes or quantity of air- or water-filled pores, hydraulic conductivity, and infiltration rate. Comparing D_b among different soils is prone to imprecise interpretation because the D_b at which root growth is limited depends on soil texture (Daddow and Warrington 1983). In general, the more coarse textured (sandy) a soil is, the higher its D_b, while the more fine textured (clayey) a soil is, the lower its D_b. Organic soils or topsoils with high organic matter content generally have the lowest values for D_b. Within a given soil, comparing one D_b value to another is can also be misleading. A large absolute increase in D_b from a relatively low value for a particular soil to a moderate value will have little effect on the properties that actually influence root growth, which are soil strength, aeration porosity, and water availability. Conversely, a small absolute increase in D_b from an already elevated value for a given soil to an even higher value will likely constitute soil damage. For example, an absolute increase in a loam D_b from 1.2 to 1.4 Mg M^{-3} (0.2 Mg m^{-3} or 17%) is larger, both in absolute and relative terms, than an increase from 1.4 to 1.5 (0.1 or 7%). Under current Forest Service standards, a 17% increase in D_b constitutes a significant impairment while the 7% increase does not. However, the increase from 1.4 to 1.5 would likely create much more growth-limiting conditions than the 17% increase. Thus, change in D_b is only useful given the initial or undisturbed value. For this reason, parameters other than bulk density are better indicators of soil function.

The interactions between soil strength, aeration porosity, and water availability have been illustrated by Letey (1985) and updated by Da Silva et al. (1994) with the creation of a single parameter, the least-limiting water range (LLWR). The LLWR has been used successfully to explain loblolly pine response to soil physical disturbance (Kelting et al. 2000), and although laborious and data intensive, could be used in some cases to monitor effects of soil physical disturbance on plant growth. Compaction increases soil strength, which becomes limiting to root growth at around 2.0 MPa (Taylor et al. 1966), although this value is species-specific. Rutting and churning tend to decrease macroporosity and hydraulic conductivity substantially, and soils with less than 10% aeration porosity are not supportive of root growth. Similarly, reductions in hydraulic conductivity can alter the surface hydrology of sites, causing shifts in a host of physical and chemical processes. Because soil type determines which of these particular properties may have greater relative influence on tree response, Aust et al. (1998) suggested that soil strength is the best indicator of damage on dry to moist soils, the decrease in aeration porosity below 10% is the best indicator of site damage on seasonally saturated soils, and the reduction of hydraulic capacity is the best indicator on frequently saturated soils.

Tree response to soil disturbance is not always a good indicator of soil function, because responses are subject to other factors, such as competing vegetation (Brais 2001). For example, compaction reduced understory competition on the Mississippi Long-Term Soil Productivity study sites. On these sites, which are moderately well-drained silt loam soils (Aquic Paleudalfs), one of the treatments applied was soil compaction at three levels: none, moderate, and severe. The moderate and severe compaction levels were induced by pulling a weighted wobble-wheel road compactor across the plot six times to achieve uniform compaction. The treatments were effective with soil bulk density of 1.3 in the uncompacted plots and 1.4 in the compacted plots (Scott et al. 2004). Planted pine biomass after 5 growing seasons was 5.9, 7.2, and 7.1 Mg ha-1 for the uncompacted, moderately compacted, and severely compacted treatments, respectively (Stagg and Scott 2006). Competing understory biomass was 5.6, 2.0, and 1.8 Mg ha-1 on the same plots, and these differences were statistically significant. Total biomass, however, was not significantly different among the compaction treatments. Furthermore, while most understory species were affected similarly, some species, such as dogwood (Cornus florida L.) and some oaks (Quercus spp.) were virtually eliminated from the compacted plots, presumably due to greater sensitivity to either increased soil strength or decreased aeration. These findings all underscore the fact that although dominant tree survival and growth is the easiest and most common bioassay of soil disturbance, all plants have individual responses to soil properties and processes (Burns and Honkala 1990), and whereas one plant may not respond negatively to a given change in soil properties or processes, others may be negatively impacted.

In rare cases, soil disturbance can create soil conditions that are actually more conducive to tree growth. In situations where a site is characterized by coarse-textured or very loosely packed soils, water holding capacity is often the soil property influencing tree growth. On these soil types, compaction can increase micropores by reducing the size of macropores. In this case, even though overall aeration may decrease, water holding capcity can be increased. This has been shown most definitively by Gomez et al. (2002) in ponderosa pine forests in California, but the phenomenon has been described in eastern forests as well (Brais 2001). Clearly, this phenomenon is very site specific and careful planning and site evaluations must be made before management prescriptions involving soil compaction are developed.

Compaction and other physical soil disturbances may impact soil functions other than tree growth. Surface compaction reduces infiltration, which increases runoff and the potential for erosion. However, MFTs rarely cause erosion and sediment transport except on areas where the forest floor is removed, such as on main skid trails and roads. A recent review of water quality research from the eastern U.S. summarized that whereas MFTs increased disturbance and water yield, measurable increases in sediment and nutrients are slight, especially where Best Management Practices (BMPs) are employed to limit the amount of bare soil created (Aust and Blinn 2004). Similarly, rutting can obstruct surface drainage and rutting and churning can reduce hydraulic capacity, thereby impeding drainage. These treatments impair better-drained soils more relative to inherently poorly drained soils (Aust et al. 1995).

3.3 Mechanical fuels treatments and soil chemical properties and processes

Organic matter disruption or removal affects a number of soil properties and cycling processes. The most direct impact of forest fuel removal is in the direct removal of C and nutrients from the forested site. The factors that govern the cumulative removal of C and nutrients from a site include the frequency of removals, the intensity of harvest or removal at each entry, the species and age of the plants being removed, and even the season of year. In general, multiple entries over a rotation or an equivalent length of time, such as with frequent selection cutting cycles or multiple thinnings, remove a greater quantity of nutrients and organic matter than single-entry harvests (even to include clearcuts), over the same length of time, and thus, harvest intensity is clearly a determinant of nutrient removal (Freedman 1981). Leaves, branches, and bark represent about 70% of the above-ground nutrients held in mature trees, and these materials represent an even greater percentage in smaller trees (Mann et al. 1998). Younger plants generally have much higher nutrient concentrations than older plants. Finally, the season of the year controls the quantity of nutrients held in the foliage. For example, newly flushed leaves in the spring have greater overall nutrient content, and senescent leaves in the fall have decreased nutrient content as trees translocate nutrients to belowground storage pools. Additionally, even in conifers the total amount of foliage in tree crowns varies by season (peaking summer and lowest in winter). Although these factors are known to control plant growth and other soil functions, some uncertainty remains as to the conditions under which removal of these materials may result in negative effects on soil function.

Concerns over harvesting and nutrient removal in eastern forests began in the early 1970s as a result of the work of Bormann and Likens (1968), who showed increased nutrient loss following clearcut harvesting, and Keeves (1966), who documented losses in productivity in the second rotation of pines in Australia on nutrient deficient soils. Interest increased dramatically in the late 1970s during the energy crisis when whole-tree harvesting was first being considered to provide biomass for energy. From these three sources of interest, a number of experiments across the eastern U.S. were designed to determine the potential nutrient loss from harvesting and other MFTs.

The general nature of these nutrient loss experiments was regional due to the regional management systems that were in place at the time. In the north-central region (210 – Warm Continental) these concerns generally were related to the effects of whole-tree harvesting on soil fertility and subsequent growth, whereas studies in the south were mostly focused on harvesting and effects of subsequent site preparation practices on soil nutrient availability and pine growth. In the northeast (M210 – Warm Continental Mountains), and in less-intensively managed forests in the south (M220 – Hot Continental Mountains), studies have focused on direct effects of whole-tree harvesting removals as well as the potential for increased leaching losses following harvest. Finally, many of the northeastern studies also were used to study the interactive processes related to harvest-caused losses and the losses and gains associated with acid precipitation. To further address these issues in a systematic way, the Long-Term Soil Productivity (LTSP) program was installed in the 1990s in a variety of locations across the southern and north-central regions to examine both harvest intensity and forest floor removal.

Harvesting, especially clearcut harvesting of entire trees (whole-tree harvesting), removes large quantities of nutrients from a site (Freedman 1981, Kimmins et al. 1985, Powers et al. 2005). Recent reviews of long-term soil C and N responses to harvesting have shown little evidence that harvesting, even whole-tree harvesting, reduces soil C and N (Knoepp and Swank

1997, Johnson and Curtis 2001, Johnson et al. 2002). These reviews were mostly centered in eastern forests; Knoepp and Swank 1997 reviewed harvesting studies in five watersheds in the southern Appalachians, Johnson and Curtis (2001) did a worldwide metanalysis of 26 studies, of which 11 were from the eastern U.S., and Johnson et al. (2002) resampled five long-term studies in a variety of southeastern ecosystems. Recent evidence from the LTSP program indicate only slight decreases in soil C through five years since harvest in Louisiana and no decreases in North Carolina (Laiho et al. 2003), and no general decreases at 5 or 10 years post-harvest across 21 installations (including the North Carolina, Louisiana, and Mississippi locations)(Powers et al. 2005).

While much of the initial concern over harvesting-induced deficiencies dealt with C and N, later studies became concerned with other nutrients, such as calcium (Ca), magnesium (Mg), potassium (K) and phosphorus (P) depletion, especially in northeastern forests where acid precipitation promotes additional Ca and Mg losses. Federer et al. (1989) reviewed the literature on losses of these nutrients in response to harvesting across the eastern U.S. and found that total soil Mg, K, and P, may decrease only by 2%-10% in 120 years depending on site and harvest intensity. Calcium losses due to leaching and harvest removals could amount to 20-60% of total ecosystem Ca over 120 years. Huntington (2000) further reviewed the evidence from several studies in the southeastern U.S. and found that harvesting and leaching losses are likely to be in excess of supply rates via weathering and cautioned that this could have a widespread (>50% of forested area) impact on productivity. Yanai et al. (2005) recently showed that apatite, a Ca-bearing soil mineral found in soils with granitic parent materials, is capable of maintaining soil Ca on many sites previously thought sensitive to depletion, but noted that soils with sedimentary parent materials may not have adequate supply rates of Ca to maintain current levels of productivity.

Harvesting-induced P removals have also been linked to reduced P availability and to growth declines. Yanai (1998) showed that whole-tree harvesting doubled the P removed compared to a similar bole-only harvested site and that harvesting reduced soil P net mineralization by 40-70% compared to an uncut control. Scott et al. (2004) indicated that whole-tree harvest and whole-tree harvest followed by forest floor removal had reduced extractable P compared to bole-only harvesting by 23% on the Louisiana and Texas LTSP locations, while having no effect on extractable P on the North Carolina or Mississippi sites. Scott and Dean (2006) and Scott et al. (2007) linked loblolly pine productivity declines caused by whole-tree harvesting compared to bole-only harvesting to the quantity of extractable P prior to harvest at the LTSP sites in Louisiana, Mississippi, and Texas.

In addition to nutrients removed in harvested material, forest floor removal can occur on trafficked areas and as part of site preparation actions such as windrowing and root raking. Forest floor displacement has been conclusively linked to nutrient loss and productivity declines (Riekirk et al. 1981, Morris et al. 1983, Stone et al. 1999, Pye and Vitousek 1985, Conde et al. 1986, Tew et al. 1986, Fox et al. 1989, Gaskin et al. 1989), and is the primary cause of erosion and sediment losses from skid trails and landings in managed forests (Aust and Blinn 2004).

3.4 Mechanical Fuels treatment and soil biological properties and processes

MFTs affect soil biological functions both through physical effects on soil properties and processes and on impacts to organic matter and chemistry, but responses are quite variable.

Because of this extreme variability and complicated nature, few generalized statements can be made regarding MFTs and biological processes and properties.

Biological activity, commonly measured through CO_2 evolution, N mineralization, or enzyme assays, is usually affected indirectly by MFTs more than directly. Biological activity is dependent on both substrate and environment, and these two factors are altered by MFTs, as discussed above. Reducing forest cover warms soil, which to a point will increase biological activity. Reduced evapotranspiration increases soil water content, which generally increases activity. If sites become waterlogged or if aeration is reduced by MFTs, activity is decreased. These basic processes have been described in many ecosystems and forests and a detailed review is beyond the scope of this paper.

In general, biological responses to MFTs have occurred where the organisms specifically use the forest floor as habitat or are particularly sensitive to soil climatic conditions, such as reduced aeration and soil temperature. Earthworm activity on the Missouri shortleaf pine-oak LTSP sites was reduced by compaction but unaffected by forest floor removal. Forest floor removal had little impact on earthworm abundance or biomass, but compaction reduced the density of *Diplocardia orrzata*, which is about 5mm in diameter, while the density of *D. smithii*, which is about 2 mm in diameter density increased (Jordan et al. 1999).

Microbial communities varied little in functional diversity with respect to either compaction or forest floor removal in Louisiana and North Carolina (230 – Subtropical) loblolly pine-dominated sites (Busse et al. 2006). Li et al. (2004) found that microbial biomass and diversity varied more within a single research site on two similar soil series (two adjacent series on a catena) than in response to compaction and forest floor removal.

Biological activity is clearly affected by soil disturbances caused by MFTs, but response are not consistent across treatments or soil types. Compaction reduced microbial biomass N in a subtropical pine site (Li et al. 2003), but N mineralization was not affected due to changes in soil climate. Soil CO_2 efflux was not affected by compaction or intensive harvesting on temperate hardwood sites in Missouri (Ponder 2005) at age 4 or by intensive harvesting on a subtropical pine site at age 10 (Butnor et al. 2006). Li et al. (2003) found that although compaction reduced N mineralization in years 2 and 5 after treatment in subtropical pine stands in North Carolina, harvest intensity had no effect on N mineralization, and that the within-site differences in soil water content on the two soil types caused the greater differences in N mineralization than any treatment, similar to their findings for microbial biomass and diversity discussed earlier.

4. Conclusions, Limitations and Future Research

One overarching conclusion that must be drawn from this review of literature pertaining to the question of how fuel management strategies influence soils in the Eastern US, is that the responses of soils (chemical, physical, or biological) can be extremely context-dependent. In other words, depending upon the conditions under which prescribed fires or mechanical fuel treatments were conducted, the impacts on soils can be quite variable. Generally speaking, the more intense the physical disturbance (heating or consumption of forest floor for prescribed fire, or compaction or erosion in mechanical operations), the more profound and long-lived the negative effects on soils. Managers must therefore take soils into special consideration when fuel management activities are planned, with a goal of minimizing these intense perturbations as much as is practicable. The research summarized here provides a reasonable reference point for these considerations, but we have also identified several limitations to our knowledge, and we suggest that more research on the effects of fuel management on soils would be useful.

Most of the studies cited in this review were conducted at the small plot or stand scale, and therefore do not provide much insight into watershed-level effects, or cumulative effects to the watershed. Detailed, spatially explicit modeling exercises will be needed to derive estimates of how the fuel treatments discussed in this review are likely to affect whole watersheds. It is likely that any models developed to assess whole watershed-level effects of fuel treatments on soils will be parameterized with the plot level data from the studies summarized in this review, but such a modeling effort has yet to be undertaken, and this represents one major avenue for future research.

Degradation of soil aggregate structure as a potential physical effect of prescribed fires has been hypothesized for Oak savanna ecosystems of Missouri, but this phenomenon has yet to be directly measured (Rhoades et al. 2004). These authors suggested that destruction of aggregate structure might partially explain observations of slow recovery of plant communities in soils where logs had 'burned out' in a prescribed fire. Such aggregate destruction may be related to the changes in soil texture, and more work on the dynamics of soil aggregate formation and stability will be needed to fully evaluate the effects of prescribed fires on soils in eastern North America.

Fire effects on roots in eastern forests is an area for much future research. Root work is tedious and time-consuming, but the potential effects of fires on root dynamics, and the attendant effects on landscape scale C sequestration make this a critical issue for researchers and forest managers to understand.

Central questions as outlined in Czimczik and Masiello (2007) surrounding the behavior and processing of "black" carbon in frequently burned soils constitute another area where a good deal of research remains to be conducted. Major areas of uncertainty include questions about how this material varies in chemical composition when formed under different combustion conditions, how BC moves into the soil profile (bioturbation or via water infiltration), how BC influences water quality, whether BC enters the dissolved fraction of SOC, whether microbial communities evolve to process this material, and whether the particle size of BC material affects any or all of the above processes. Overall, this and other aspects of how prescribed fire influences the carbon balance of forested ecosystems in the eastern US would benefit from a much more detailed accounting than is currently available.

Although soil biota – both macro and microarthropods – have been demonstrated to have substantial effects on soil processes in agricultural (and some forested) ecosystems of the eastern US, their responses to fuel management practices are not well known. More work examining the responses of the soil invertebrate community to both prescribed fire and mechanical fuel treatments would allow for better understanding of how these management activities influence the functioning of soils when these techniques are applied.

Nearly all of the soil responses to fuel treatments discussed in this chapter have some temporal dimension that is extremely difficult to evaluate in short term studies. Further complications arise from the fact that different soil functional responses to fire (e.g. nutrient mineralization rate vs. soil organic matter loss or accrual) will take different amounts of time to be evident. In other words, some responses of soil ecosystems may be clear in a year or two following fire, but others may take decades to reach equilibrium. Fortunately, the US Forest Service (among other research organizations) is committed to maintaining long-term studies including soils-based studies (e.g., studies at the Experimental Forests and co-located LTER sites, and the LTSP plots described above). Such long-term experimentation will be critical to guiding the management of natural resources (including soil) in the future, and data from such studies will be of great value when models are developed to fully address these issues at the landscape scale (Richter et al. 2008).

Literature Cited

Alexis, M.A., D.P. Rasse, C. Rumpel, G. Bardoux, N. Péchot, P. Schmalzer, B. Drake, and A. Mariotti. 2007. Fire impact on C and N losses and charcoal production in a scrub oak ecosystem. Biogeochemistry 82: 201-216.

Aust W.M. and C.R. Blinn. 2004. Forestry best management practices for timber harvesting and site preparation in the eastern United States: an overview of water quality and productivity research during the past 20 years (1982-2002). Water, Air, and Soil Pollution: Focus 4:5-36.

Aust W.M., J.A. Burger, E.A. Carter, D.P. Preston, and S.C. Patterson. 1998. Visually determined soil disturbance classes used as indices of forest harvesting disturbance. Southern Journal of Applied Forestry 22:245-250.

Aust W.M., M.D. Tippett, J.A. Burger, and W.H. McKee_Jr. 1995. Compaction and rutting during harvesting affect better drained soils more than poorly drained soils on wet pine flats. Southern Journal of Applied Forestry 19:72-77.

Bates P.C., C.R. Blinn, and A.A. Alm. 1993. Harvesting impacts on quaking aspen regeneration in northern Minnesota. Canadian Journal of Forest Research 23:2403-2412.

Benning T.L., and T.R. Seastedt. 1997. Effects of fire, mowing and nitrogen additions on root characteristics in tallgrass prairie. Journal of Vegetation Science 8:541-546.

Boerner, R.E.J., T.A. Waldrop, and V.B. Shelburne. 2006. Wildfire mitigation strategies affect soil enzyme activity and soil organic carbon in loblolly pine (*Pinus taeda*) forests. Canadian Journal of Forest Research 36:3148-3154.

Boerner, R.E.J., J.A. Brinkman, and A. Smith. 2005. Seasonal variations in enzyme activity and organic carbon in soil of a burned and unburned hardwood forest. Soil Biology and Biochemistry 37: 1419-1426.

Boerner, R.E.J., J. Huang, and S.C. Hart. 2008. Fire, thinning, and the carbon economy: Effects of fire and fire surrogate treatments on estimated carbon storage and sequestration rate. Forest Ecology and Management 255: 3081-3097.

Boring, L.R., J.J. Hendricks, C.A. Wilson, and R.J. Mitchell. 2004. Season of burn and nutrient losses in a longleaf pine ecosystem. International Journal of Wildland Fire 13: 443-453.

Bormann F.H. and G.E. Likens. 1968. Nutrient loss accelerated by clear-cutting of a forest ecosystem. Science 159:882-884.

Brais S. 2001. Persistence of soil compaction and effects on seedlings growth in northwestern Quebec. Soil Science Society of America Journal 65:1263-1271.

Brand, R.H. 2002. Effect of prescribed burning on epigeic springtails (Insecta: Collembola) of woodland litter. American Midland Naturalist 148:383-393.

Brockway, D.G., and C.E. Lewis. 1997. Long-term effects of dormant-season prescribed fire on plant community diversity, structure and productivity in a longleaf wiregrass ecosystem. Forest Ecology and Management 96:167-183.

Brye, K.R., 2006. Soil physiochemical changes following 12 years of annual burning in a humid-subtropical tallgrass prairie: a hypothesis. Acta Oecologica 30: 407-413.

Burger, J.A. 1994. Cumulative effects of silvicultural technology on sustained forest productivity. pp. 59-70, In: (Mahendrappa, M.K. and C.M.S.C.T. Simpson, eds.) Assessing the Effects of Silvicultural Practices on Sustained Productivity: A Proceedings of the IEA/BE Workshop '93. Canadian Forest Service - Maritimes Region, Natural

Resources Canada, P.O. Box 4000, Fredericton, N.B. E3B 5P7 , IEA/BA Task IX Activity 4 Report 3. Information Report M-X-191.

Burns, R.M. and B.H. Honkala. 1990. Silvics of North America. Vol 1. Conifers. Vol 2.

Busse, M.D., S.E. Beattie, R.F. Powers, F.G. Sanchez, and A.E. Tiarks. 2006. Microbial community responses in forest mineral soil to compaction, organic matter removal, and vegetation control. Canadian Journal of Forest Research 36:577-588.

Hardwoods. Washington, DC. Agriculture Handbook 654.

Butnor, J.R., K.H. Johnsen, and F.G. Sanchez. 2006. Whole-tree forest floor removal from a loblolly pine plantation have no effect on forest floor CO2 efflux 10 years after harvest. Forest Ecology and Management 227:89-95.

Callaham, M.A., Jr., J.M. Blair, T.C. Todd, D.J. Kitchen, and M.R. Whiles. 2003. Macroinvertebrates in North American tallgrass prairie soils: Effects of fire, mowing, and fertilization on density and biomass. Soil Biology and Biochemistry 35:1079-1093.

Callaham, M.A., Jr., P.H. Anderson, T.A. Waldrop, D.J. Lione, and V.B. Shelburne. 2004. Litter decomposition and soil respiration responses to fuel-reduction treatments in Piedmont loblolly pine forests . *In* K. Connor, editor. Proceedings, 12th Southern Silvicultural Research Conference, Biloxi, MS. Gen. Tech. Rep. SRS–71. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. Pp. 25-29.

Carter M.C., T.J. Dean, M. Zhou, M.G. Messina, and Z. Wang. 2002. Short-term changes in soil C, N, and biota following harvesting and regeneration of loblolly pine (Pinus taeda L.). Forest Ecology and Management 164:67-88.

Coleman, D.C., D.A. Crossley, Jr., and P.F. Hendrix. 2004. Fundamentals of Soil Ecology. Academic Press, Inc., San Diego, CA. 386 pp.

Coleman, T.W., and L.K. Rieske. 2006. Arthropod response to prescription burning at the soil-litter interface in oak-pine forests. Forest Ecology and Management 233: 52-60.

Conde L.F., B.F. Swindel, and J.E. Smith. 1986. Windrowing affects early growth of slash pine. Southern Journal of Applied Forestry 10:81-84.

Czimczik, C.I. and C.A. Masiello. 2007. Controls on black carbon storage in soils. Global Biogeochemical Cycles 21: GB3005, doi:10.1029/2006GB002798.

Dai, X., T.W. Boutton, B. Glaser, R.J. Ansley, and W. Zech. 2005. Black carbon in a temperate mixed-grass savanna. Soil Biology and Biochemistry 37: 1879-1881.

Daddow, R.L. and G.E. Warrington. 1983. Growth-limiting bulk densities as affected by soil texture. pp.21 pp. In: Watershed Development Group, Fort Collins, CO.Rep. TN-00005.

Da Silva A.P., B.D. Kay, and E. Perfect. 1994. Characterization of the least limiting water range of soils. Soil Science Society of America Journal 58:1775-1781.

DeBano, L.F. 2000. The role of fire and soil heating on water repellency in wildland environments: a review. Journal of Hydrology 231-232: 195-206.

DeLuca, T.H. and G.H. Aplet. 2008. Charcoal and carbon storage in forest soils of the Rocky Mountain west. Frontiers in Ecology and the Environment 6:18-24.

Dijkstra, F.A., K. Wrage, S.E. Hobbie, and P.B. Reich. 2006. Tree patches show greater N losses but maintain higher soil N availability than grassland patches in a frequently burned oak savanna. Ecosystems 9: 441-452

Dress, W.J., and R.E.J. Boerner. 2004. Patterns of microarthropod abundance in oak-hickory forest ecosystems in relation to prescribed fire and landscape position. Pedobiologia 48:1-8.

Dyrness C.T. 1965. Soil surface condition following tractor and high-lead logging in the Oregon

Cascades. Journal of Forestry 53:272-275.

Elliot, K.J., J.M. Vose, and B.D. Clinton. 2002. Growth of eastern white pine (*Pinus strobus* L.) related to forest floor consumption by prescribed fire in the southern Appalachians. Southern Journal of Applied Forestry 26:18-25.

Farrish K.W., J.C. Adams, and C.G. Vidrine. 1995. Survival and growth of planted loblolly pine seedlings on a severely rutted site. Tree Planters' Notes 46:28-31.

Federer C.A., J.W. Hornbeck, L.M. Tritton, C.W. Martin, and R.S. Pierce. 1989. Long-term depletion of calcium and other nutrients in eastern US forests. Environmental Management 13:593-601.

Foil R.R. and C.W. Ralston. 1967. The establishment and growth of loblolly pine seedlings on compacted soils. Soil Science Society of America Journal 31:565-568.

Fox, T. R. 2000. Sustained productivity in intensively managed forest plantations. Forest Ecology and Management 138:187-202.

Fox, T.R., L.A. Morris, and R.A. Maimone. 1989. The impact of windrowing on the productivity of a rotation age loblolly pine plantation. pp.133-140 , In: (Miller J.H., eds.) Proceedings of the Fifth Biennial Southern Silvicultural Research Conference. USDA Forest Service, Southern Forest Experiment Station, New Orleans, LA.General Technical Report SO-74.

Freedman, B. 1981. Intensive forest harvest: a review of nutrient budget considerations. Info. Rept. M-X-121. Maritimes For. Res. Centre, Can. For. Serv., Dept. Environ., Fredericton, N.B. 78p.

Gaskin J.W., W.L. Nutter, and T.M. McMullen. 1989. Comparison of nutrient losses by harvesting and site preparation practices in the Georgia Piedmont and Coastal Plain. Georgia Forest Research Report 77. 7 p.

Giai, C., and R.E.J. Boerner. 2007. Effects of ecological restoration on microbial activity, microbial functional diversity, and soil organic matter in mixed-oak forests of southern Ohio, USA. Applied Soil Ecology 35: 281-290.

Glasgow, L.S., and G.R. Matlack. 2007. Prescribed burning and understory composition in a temperate deciduous forest, Ohio, USA. Forest Ecology and Management 238: 54-64.

Glitzenstein, J.S., D.R. Streng, and D.D. Wade. 2003. Fire frequency effects on longleaf pine (*Pinus palustris*, P. Miller) vegetation in South Carolina and northeast Florida, USA. Natural Areas Journal 23:22-37.

Gomez A., R.F. Powers, M.J. Singer, and W.R. Horwath. 2002. Soil compaction effects on growth of young ponderosa pine following litter removal in California's Sierra Nevada. Soil Science Society of America Journal 66:1334-1343.

Gray, D.M., and J. Dighton. 2006. Mineralization of forest liter nutrients by heat and combustion. Soil Biology and Biochemistry 38: 1469-1477.

Greacen, E. L. and R. Sands. 1980. Compaction of forest soils. A review. Australian Journal of Soil Research 18: 163-189.

Grigal D.F. 2000. Effects of extensive forest management on soil productivity. Forest Ecology and Management 138:167-185.

Hanula, J.L. and D.D. Wade. 2003. Influence of long-term dormant season burning and fire exclusion on ground-dwelling arthropod populations in longleaf pine flatwoods ecosystems. Forest Ecology and Management 175:163-184.

Hatchell G.E. 1981. Site preparation and fertilizer increase pine growth on soils compacted in logging. Southern Journal of Applied Forestry 5:79-83.

Hatchell G.E., C.W. Ralston, and R.R. Foil. 1970. Soil disturbances in logging. Journal of

Forestry 68:772-775.

Haywood, J.D. 2007. Influence of herbicides and felling, fertilization, and prescribed fire on longleaf pine establishment and growth through six growing seasons. New Forests 33: 257-279.

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

Henig-Sever, N., D. Poiakov, and M. Broza. 2001. A novel method for estimation of wildfire intensity based on ash pH and soil microarthropod community. Pedobiologia 45:98-106.

Huang, J., and R.E.J. Boerner. 2007. Effects of fire alone or combined with thinning on tissue nutrient concentrations and nutrient resorption in *Desmodium nudiflorum.* Oecologia 153: 233-243.

Hubbard, R.M., J.M. Vose, B.D. Clinton, K.J. Elliott, and J.D. Knoepp. 2004. Stand restoration burning in oak-pine forests in the southern Appalachians: effects on aboveground biomass and carbon and nitrogen cycling. Forest Ecology and Management 190: 311-321.

Huntington T.G. 2000. The potential for calcium depletion in forest ecosystems of southeastern United States: review and analysis. Global Biogeochemical Cycles 14:623-638.

Hutchinson, T.F., E.K. Sutherland, and D.A. Yaussy. 2005. Effects of repeated prescribed fires on the structure, composition and regeneration of mixed-oak forests in Ohio. Forest Ecology and Management 218:210-228.

James, S.W. 1982. Effects of fire and soil type on earthworm populations in a tallgrass prairie. Pedobiologia 24:37-40.

Jenny, H. 1941. Factors of Soil Formation. McGraw-Hill, New York.

Joergensen, J.R. and C.S. Hodges, Jr. 1970. Microbial characteristics of a forest soil after twenty years of prescribed burning. Mycologia 62:721-726.

Johnson D.W. and P.S. Curtis. 2001. Effects of forest management on soil C and N storage: meta analysis. Forest Ecology and Management 140:227-238.

Johnson D.W., J.D. Knoepp, W.T. Swank, J. Shan, L.A. Morris, D.H. Van Lear, and P.R. Kapeluck. 2002. Effects of forest management on soil carbon:results of some long-term resampling studies. Environmental Pollution 116:201-208.

Jordan, D., F. Li, F. Ponder, Jr., E.C. Berry, V.C. Hubbard, and K.Y. Kim. 1999. The effects of forest practices on earthworm populations and soil microbial biomass in a hardwood forest in Missouri. Applied Soil Ecology 13:31-38.

Jordan D., F. Ponder Jr., and V.C. Hubbard. 2003. Effects of soil compaction, forest leaf litter and nitrogen fertilizer on two oak species and microbial activity. Applied Soil Ecology 23:33-41.

Kalisz, P.J., and J.E. Powell. 2000. Effects of prescribed fire on soil invertebrates in upland forests on the Cumberland Plateau of Kentucky, USA. Natural Areas Journal 20:336-341.

Keeves, A. 1966. Some evidence of loss of productivity with successive rotations of *Pinus radiata* in the south east of south Australia . *Australian Forestry* 30:57-63.

Kelting D.L., J.A. Burger, and S.C. Patterson. 2000. Early loblolly pine growth response to changes in the soil environment. New Zealand Journal of Forestry Science 30:206-224.

Kimmins, J.P., D. Binkley, L. Chatarpaul, and J. de Catanzaro. 1985. Biogeochemistry of temperate forest ecosystems: literature on inventories and dynamics of biomass and nutrients. Can. For. Serv., Nat. For. Inst., Petawawa, Ont. Information Rept. PI-X-47 E/F 227pp.

King, T. and S. Haines. 1979. Soil compaction absent in plantation thinning. Research Note SO-251. Southern Forest Experiment Station, New Orleans, LA. 4 p.

Kitchen, D.J, J.M. Blair, and M.A. Callaham, Jr. *In press*. Annual fire and mowing alter biomass, depth distribution, and C and N content of roots and soil in tallgrass prairie. Plant and Soil. XX:xxx-xxx.

Knapp A.K., J.M. Briggs, J.M. Blair, and C.L. Turner. 1998. Patterns and controls of aboveground net primary production in tallgrass prairie. In: Knapp A.K., J.M. Briggs, D.C. Hartnett, and S.L. Collins (eds), Grassland dynamics: long-term ecological research in tallgrass prairie. Oxford University Press, New York, pp 193-221.

Knoepp J.D. and W.T. Swank. 1997. Forest management effects on surface soil carbon and nitrogen. Soil Science Society of America Journal 61:928-935.

Laiho R., F. Sanchez, A.E. Tiarks, P.M. Dougherty, and C.C. Trettin. 2003. Impacts of intensive forestry on early rotation trends in site carbon pools in the southeastern US. Forest Ecology and Management 174:177-189.

Lajeunesse, S.D., J.J. Dilustro, R.R. Sharitz, and B.S. Collins. 2006. Ground layer carbon and nitrogen cycling and legume nitrogen inputs following fire in mixed pine forests. American Journal of Botany 93(1): 84-93.

Li, Q.C., H.L. Allen, and C.A. Wilson. 2003. Nitrogen mineralization dynamics following the establishment of a loblolly pine plantation. Canadian Journal of Forest Research 33:364-374.

Li, Q.C., H.L. Allen, and A.G. Wollum, II. 2004. Microbial biomass and bacterial functional diversity in forest soils: effects of organic matter removal, compaction, and vegetation control. Soil Biology and Biochemistry 36:571-579.

Liechty, H.O., M.G. Sheldon, K.R. Luckow, D.J. Turton. 2002. Impacts of shortleaf pine-hardwood forest management on soils in the Ouachita Highlands: A review. Southern Journal of Applied Forestry. 26:43-51.

Liechty, H.O., K.R. Luckow, and J.M. Guldin. 2005. Soil chemistry and nutrient regimes following 17-21 years of shortleaf pine-bluestem restoration in the Ouachita Mountains of Arkansas. Forest Ecology and Management 204: 345-357.

Letey, J. 1985. Relationship between soil physical properties and crop production. Adv. Soil Sci. 1:277–294.

Lockaby B.G. and C.G. Vidrine. 1984. Effect of logging equipment traffic on soil density and growth and survival of young loblolly pine. Southern Journal of Applied Forestry 8:109-112.

Lull, H.W. 1959. Soil compaction on forest and rangelands. USDA For. Serv. Misc. Pub. No 768. U.S. Gov. Print. Office, Washington, DC.

Mann, L.K., D.W. Johnson, D.C. West, D.W. Cole, J.W. Hornbeck, C.W. Martin, H. Riekerk, C.T. Smith, W.T. Swank, L.M. Tritton, and D.H. Van_Lear. 1988. Effects of whole-tree and stem-only clearcutting on postharvest hydrologic losses, nutrient capital, and regrowth. Forest Science 34:412-428.

McCarthy, D.R., and K.J. Brown. 2006. Soil respiration responses to topography, canopy cover, and prescribed burning in an oak-hickory forest in southeastern Ohio. Forest Ecology and Management 237: 94-102.

Metz, L.J. and M.H. Farrier. 1971. Prescribed burning and soil mesofauna on the Santee Experimental Forest. Pages 100-106 in USDA Forest Service, Prescribed Burning Symposium Proceedings, Charleston, SC. Southeastern Forest Experiment Station, Asheville, NC.

Miller J.H. and D.L. Sirois. 1986. Soil disturbance by skyline yarding vs. skidding in a loamy hill forest. Soil Science Society of America Journal 50:1579-1583.

Mitchell, M.L., A.E. Hassan, C.B. Davey and J.D. Gregory. 1982. Loblolly pine growth in compacted greenhouse soils. Trans. ASAE 25:304–312.

Miwa M., W.M. Aust, J.A. Burger, S.C. Patterson, and E.A. Carter. 2004. Wet-weather harvesting and site preparation effects on coastal plain sites: a review. Southern Journal of Applied Forestry 28:137-151.

Moehring D.M. and I.W. Rawls. 1970. Detrimental effects of wet weather logging. Journal of Forestry 68:166-167.

Morris L.A., W.L. Pritchett, and B. Swindel. 1983. Displacement of nutrients into windrows during site preparation of a flatwood forest. Soil Science Society of America Journal 47:591-594.

Neary, D.G., C.C. Klopatek, L.F. DeBano, P.F. Ffolliott. 1999. Fire effects on belowground sustainability: a review and synthesis. Forest Ecology and Management 122:51-71.

Neill, C., B. Von Holle, K. Kleese, K.D. Ivy, A.R. Collins, C. Treat, and M. Dean. 2007a. Historical influences on the vegetation and soils of the Martha's Vineyard, Massachusetts coastal sandplain: Implications for conservation and restoration. Biological Conservation 136: 17-32.

Neill, C., W.A. Patterson III, and D.W. Crary, Jr. 2007b. Responses of soil carbon, nitrogen and cations to the frequency and seasonality of prescribed burning in a Cape Cod oak-pine forest. Forest Ecology and Management 250: 234-243.

O'Brien, J.J., J.K. Hiers, R.J. Mitchell, J.M. Varner, III, and K. Mordecai. *In review*. Acute physiological stress from root loss drives post duff fire mortality in longleaf pine (*Pinus palustris* Mill.). Submitted to International Journal of Wildland Fire, July 2009.

Phillips, R.J. and T.A. Waldrop. 2008. Changes in vegetation structure and composition in response to fuel reduction treatments in the South Carolina Piedmont. Forest Ecology and Management 225:3107-3116.

Ponder, F. 2005. Effect of soil compaction and biomass removal on soil CO_2 efflux in a Missouri forest. Communications in Soil Science and Plant Analysis 36:1301-1311.

Pomeroy K.B. 1949. The germination and initial establishment of loblolly pine under various surface soil conditions. Journal of Forestry 47:541-543.

Poth, M., I.C. Anderson, H.S. Miranda, A.C. Miranda, and P.J. Riggan. 1995. The magnitude and persistence of soil NO, N_2O, CH_4, and CO_2 fluxes from burned tropical savanna in Brazil. Global Biogeochemical Cycles 9:503-513.

Powers, R.F., D.H. Alban, R.E. Miller, A.E. Tiarks, C.G. Wells, P.E. Avers, R.G. Cline, R.O. Fitzgerald, and N.S. Loftus_Jr. 1990. Sustaining site productivity in North American forests: problems and prospects. pp. 49-79, In: (Gessel, S.P., D.S. Lacate, G.F. Weetman, and R.F. Powers, eds.) Sustained Productivity of Forest Soils. Proceedings of the Seventh North American Forest Soils Conference. University of British Columbia, Vancouver, B.C., Canada.

Powers R.F., D.A. Scott, F.G. Sanchez, R.A. Voldseth, D. Page-Dumroese, J.D. Elioff, and D.M.

Stone. 2005. The North American long-term soil productivity experiment: Findings from the first decade of research. Forest Ecology and Management 220:31-50.

Pye J.M. and P.M. Vitousek. 1985. Soil and nutrient removals by erosion and windrowing at a southeastern U.S. Piedmont site. Forest Ecology and Management 11:145-155.

Renbuss, M., G.A. Chilvers, and L.D. Pryer. 1973. Microbiology of an ashbed. Proceedings of the Royal Linnean Society of New South Wales. 97:302-310.

Rhoades, C.C., A.J. Meier, and A.J. Rebertus. 2004. Soil properties in fire-consumed log burnout openings in a Missouri oak savanna. Forest Ecology and Management 192: 277-284.

Richter, D.DeB., M. Hofmockel, M.A. Callaham, Jr., D.S. Powlson, and P. Smith. 2007. Long-term soil experiments: keys to managing Earth's rapidly changing ecosystems. Soil Science Society of America Journal 71:266-279.

Riekirk, H., L.F. Conde, J.C. Hendrickson, and W.S. Cain. 1981. Research on environmental and site effects of forest management practices in the Lower Coastal Plain. pp.331-3338 , In: (Barnett J.P., eds.) Proceedings of the First Biennial Southern Silvicultural Research Conference. USDA Forest Service, Southern Forest Experiment Station, New Orleans, LA.General Technical Report GTR-SO-34.

Reisinger, T.W., W.M. Aust, and D.B. Powell Jr. 1993. The effects of a wet-weather thinning operation in a coastal plain loblolly pine stand. pp.251-254 , In: (Brissett J.C., eds.) Proceedings of the Seventh Biennial Southern Silvicultural Research Conference. General Technical Report SO.93. USDA Forest Service, Southern Forest Experiment Station, New Orleans, LA.

Sanchez, F.G., D.A. Scott, and K.H. Ludovici. 2006. Negligible effects of severe organic matter removal and soil compaction on loblolly pine growth over 10 years. Forest Ecology and Management 227:145-154.

Scheerer, G.A., W.M. Aust, J.A. Burger, and W.H. McKee Jr. 1994. Skid trail amelioration following timber harvests on wet pine flats in South Carolina: two-year results. pp.236-243, In: (Edwards M.B., eds.) Proceedings of the Eighth Biennial Southern Silvicultural Research Conference. General Technical Report SRS-1. U.S. Dept. Agric., For. Serv., Southern Research Station, Asheville, NC.

Scheuner, E.T., F. Makeschin, E.D. Wells, and P.Q. Carter. 2004. Short-term impacts of harvesting and burning disturbances on physical and chemical characteristics of forest soils in western Newfoundland, Canada. European Journal of Forest Research 123: 321-330.

Schlesinger. W.H. Biogeochemistry: An Analysis of Global Change. Academic Press, San Diego, 443 pp.

Scott D.A. and T.J. Dean. 2006. Energy trade-offs between intensive biomass utilization, site productivity loss, and ameliorative treatments in loblolly pine plantations. Biomass and Bioenergy 30:1001-1010.

Scott, D.A., J. Novosad, and G. Goldsmith. 2007. Ten-year results from the North American Long-Term Soil Productivity study in the western Gulf coastal plain. pp.331-340 , In: (Furniss M., C. Clifton, and K. Ronnenburg, eds.) Advancing the Fundamental Sciences: Proceedings of the Forest Service National Earth Sciences Conference. Gen. Tech. Rep. PNW-689, USDA Forest Service, Portland, OR.

Scott, D.A., A.E. Tiarks, F.G. Sanchez, M.L. Elliott-Smith, and R.H. Stagg. 2004. Forest soil productivity on the southern long-term soil productivity sites at age 5. pp.372-377 , In: (Connor K.F., eds.) Proceedings of the 12th Biennial Southern Silvicultural Research Conference. USDA Forest Service, Southern Research Station, Asheville, NC.General Technical Report SRS-71.

Seastedt, T.R. 1984. Microarthropods of burned and unburned tallgrass prairie. Journal of the Kansas Entomological Society 57:468-476.

Siegel-Issem C.M., J.A. Burger, R.F. Powers, F. Ponder, and S.C. Patterson. 2005. Seedling root growth as a function of soil density and water content. Soil Science Society of America Journal 69:215-226.

Simmons, G.L. and A.W. Ezell. 1982. Root development of loblolly pine seedlings in compacted soils. pp.26-29 , In: (Jones Jr. E.P., eds.) Proceedings of the Second Biennial Southern Silvicultural Research Conference. USDA Forest Service, Southeastern Forest Experiment Station, Asheville, NC.General Technical Report GTR.SE.24.

Simmons, G.L. and P.E. Pope. 1985. Effects of soil compaction on root growth characteristics of yellow-poplar and sweetgum seedlings. pp. 264-268, In: (Dawson, J.O. and K.A. Majerus, eds.) Proceedings of the Fifth Central Hardwood Forest Conference. Society of American Foresters Publication 85-05, Bethesda, MD.

Smidt M.F. and C.R. Blinn. 2002. Harvest caused soil disturbance decreased suckering capacity of quaking aspen (Populus tremuloides Michx.) following growing season harvests in Minnesota, USA. Forest Ecology and Management 163:309-317.

Staddon, W.J., L.C. Duchense, and J.T. Trevors. 1998. Acid phosphatase, alkaline phosphatase and arylsulfatase activities in soils from a jack pine *(Pinus banksiana* Lamb.) ecosystem after clear-cutting, prescribed burning, and scarification. Biology and Fertility of Soils 27: 1-4.

Stagg, R.H. and D.A. Scott. 2006. Understory growth and composition resulting from soil disturbances on the long-term soil productivity study sites in Mississippi. pp.52-56 , In: (Connor K.F., eds.) Proceedings of the 13th Biennial Southern Silvicultural Research Conference. USDA Forest Service, Southern Research Station, Asheville, NC.General Technical Report SRS-92.

Steber A., K. Brooks, C.H. Perry, and R. Kolka. 2007. Surface compaction estimates and soil sensitivity in aspen stands of the Great Lakes States. Northern Journal of Applied Forestry 24:276-281.

Stone, D.M. 2002. Lake states aspen productivity following soil compaction and organic matter removal. pp.59-67 , In: (Parker S. and S. S. Hummel, eds.) Beyond 2001: A Silvicultural Odyssey to Sustaining Terrestrial and Aquatic Ecosystems Proceedings of the 2001 National Silviculture Workshop, JUIY 2002 May 6-1 0, Hood River, Oregon. USDA Forest Service, Pacific Northwest Research Station, Portland, OR.General Technical Report GTR-PNW-546.

Stone D.M. and J.D. Elioff. 2000. Soil disturbance and aspen regeneration on clay soils: three case histories. Forestry Chronicle 76:747-752.

Stone, D.M., J.A. Gates, and J.D. Elioff. 1999. Are we maintaining aspen productivity on sand soils? pp. 177-184, In: (Ek, A.R. and B. ZumBahlen, eds.) Improving forest productivity for timber...A key to sustainability. Department of Forest Resources, University of Minnesota, St. Paul, MN.

Swift, L.W., K.J. Elliott, R.D. Ottmar, and R.E. Vihnanek. 1993. Site preparation burning to improve southern Appalachian pine-hardwood stands – site characteristics and soil-erosion, moisture, and temperature. Canadian Journal of Forest Research 23:2242-2254.

Sword Sayer, M.A., and J.D. Haywood. 2006. Fine root production and carbohydrate concentrations of mature longleaf pine (*Pinus palustris* P. Mill.) as affected by season of prescribed fire and drought. Trees 20: 165-175.

Taylor, H.M., G.M. Roberson, J.J. Parker Jr. 1966. Soil strength-root penetration relations for medium- to coarse-textured soil materials. Soil Sci. 102:18-22.

Tellier, R., L.C. Duchesne, J.C. Ruel, and R.S. McAlpine. 1995. Effects of directed burning intensity on diversity of plant-species in a jack pine stand (*Pinus banksiana* Lamb). Ecoscience 2: 159-167.

Tew D.T., L.A. Morris, H.L. Allen, and C.G. Wells. 1986. Estimates of nutrient removal, displacement and loss resulting from harvest and site preparation of a *Pinus taeda* plantation in the Piedmont of North Carolina. Forest Ecology and Management 15:257-267.

Tiarks, A. E. 1990. Growth of slash pine planted in soil disturbed by wet-weather logging. Journal of Soil and Water Conservation 45: 405-407.

Tworkoski T.J., J.A. Burger, and D.W. Smith. 1983. Soil texture and bulk density affect early growth of white oak seedlings. Tree Planters' Notes 34:22-25.

Van Lear, D.H., J.E. Douglass, S.K. Cox, and M.K. Augspurger. 1985. Sediment and nutrient export in runoff from burned and harvested pine watersheds in the South Carolina Piedmont. Journal of Environmental Quality 14:169-174.

Wardle, D.A., R.D. Bardgett, J.N. Klironomos, H. Setälä, W.H. van der Putten, and D.H. Wall.

2004. Ecological linkages between aboveground and belowground biota. Science 304:1629-1633.

Wardle, D.A., M.-C. Nilsson, and O. Zackrisson. 2008. Fire-derived charcoal causes loss of forest humus. Science 320: 629.

Xu Y.-J., J.A. Burger, and W.M. Aust. 2000. Responses of surface hydrology and early loblolly pine growth to soil disturbance and site preparation in a lower coastal plain wetland. New Zealand Journal of Forestry Science 30:250-265.

Yanai R.D. 1998. The effect of whole-tree harvest on phosphorus cycling in a northern hardwood forest. Forest Ecology and Management 104:281-295.

Yanai R.D., J.D. Blum, S.P. Hamburg, M.A. Arthur, C.A. Nezat, and T.G. Siccama. 2005. New insights into calcium depletion in northeastern forests. Journal of Forestry 103:14-20.

Zeleznik, J.D., and D.I. Dickmann. 2004. Effects of high temperatures on fine roots of mature red pine (*Pinus resinosa)* trees. Forest Ecology and Management 199: 395-409.

Zenner E.K., J.T. Fauskee, A.L. Berger, and K.J. Puettmann. 2007. Impacts of skidding traffic intensity on soil disturbance, soil recovery, and aspen regeneration in North Central Minnesota. Northern Journal of Applied Forestry 24:177-183.

Table 1. Summary of physical effects of prescribed fire, or treatments including prescribed fire (e.g. thinning, or herbicide treatment in combination with Rx fire) on soils in eastern North America.

Fire Management Objective	Ecosystem	Location (State/ Province)	Ecological Division	O horizon	Moisture	Temperature	Texture	Bulk density	Citation
Habitat	Outwash sandplain	MA	220					=	Neill et al. 2007a
	Scrub oak	FL	230	+ charcoal					Alexis et al. 2007
Restoration	Oak	IL	220	- mass					Brand 2002
	Oak savanna	MO	220				+ clay particles	=	Rhoades et al. 2004
	Oak/hickory	OH	220		=	=			McCarthy and Brown 2006
	Oak/hickory	OH	220	- mass					Dress and Boerner 2004
	Oak/hickory	TN	M220		=				Jackson et al. 2006
	Oak/hickory – grass	KY	220		-	+			Rhoades et al. 2002
	Oak/pine	MA	220	- summer burns				+ in O-horizon	Neill et al. 2007b
	Oak/pine	TN/GA	M220	- mass Oi only	+ short term	= at 10 cm			Hubbard et al. 2004
Site prep	Shortleaf pine – grass	AR	M230	Site dependent - or =					Liechty et al. 2002
	Black spruce	NF	210?	- in Oi and Oa					Scheuner et al. 2004
	White Pine	NC	M220	- mass Oi					Vose and Swank 1993
	White pine	NC	M220		+ in fell/burn	+ in fell/burn			Swift et al. 1993 Swift et al. 1993

Notes: Ecological Division labels are consistent with those presented by Cleland et al. (2005), and Chapter 1 of this volume. A '+' indicates a statistically significant increase in the measured variable, '=' indicates no statistically significant change in the variable, and '-' indicates a statistically significant decrease in the variable following prescribed fire or treatments including prescribed fire. For O-horizon column symbols are consistent with term in US Soil Taxonomy (Oi = litter layer; Oa = Fermentation layer; Oe = humus layer), "- m ass" indicates that the mass of the O horizon was significantly reduced.

Table 2. Summary of selected soil chemical effects of prescribed fire, or treatments including prescribed fire (e.g. thinning, or herbicide treatment in combination with Rx fire) in eastern North America.

Fire Management Objective	Ecosystem	Location (State/ Province)	Ecological Division	C	C (black)	N (total)	C:N	NH_4	NO_3	N_{min}	DON	Citation
Fuel reduction	Many	Many	220, M220, 230	=								Boerner et al. 2008
	Oak savanna	MN	210			-		+	=	+	+	Dijkstra et al. 2006
	Oak/Hickory	OH	220	=				=	=		=	Giai and Boerner 2007
	Pine barrens	NJ	230			-		+	+			Gray and Dighton 2006
	Prairie	AR		+		+	+			+		Brye 2006
Habitat	Scrub oak	FL	230	-/=	+	-/=						Alexis et al. 2007
	Longleaf pine	GA	230	-		-		+	=			Boring et al. 2004
	Outwash sandplain	MA	220	=		=	=	=	=	=		Neill et al. 2007
Restoration	Grassland	MD		=								Sherman et al. 2005
	Oak/Hickory	OH	220	+						=/-		Boerner et al. 2005
	Oak savanna	MO	220	=		=	+	=	+			Rhoades et al. 2004
	Oak/Hickory	OH	220									Huang & Boerner 2007
	Oak/Pine	MA	220	=		=	=					Neill et al. 2007
	Oak/Pine	TN/GA	M220	-/=		-/=		=	=			Hubbard et al. 2004
	Pine-bluestem	AR	M230	+		+	+					Leichty et al. 2005
	Prosopis savanna	TX	230?	=	+							Dai et al. 2005
	Oak/Hickory - grass barrens	KY	220					+	=	+		Rhoades et al. 2002
Site prep	White pine	NC	M220			=		+	=/+	+/=		Knoepp and Swank 1993
	White pine	NC	M220	=		=		+	=/+	+/=		Knoepp et al. 2004
	White Pine	NC	M220	-/=		-/=						Vose and Swank 1993
	Jack pine	ON	210?									Staddon et al. 1998
	Black spruce	NF	210?	=		=	=					Scheuner et al. 2004

Notes: Ecological Division identifiers are consistent with those presented by Cleland et al. 2005, and Chapter 1 of this volume. A '+' indicates a statistically significant increase in the measured variable, '=' indicates no statistically significant change in the variable, and '-' indicates a statistically significant decrease in the measured variable following prescribed fire or treatments including prescribed fire. When two symbols are presented for a given variable, the first symbol represents a shallower soil depth (typically O-horizon soil), and the second symbol represents a deeper soil depth.

Table 3. Summary of additional selected soil chemical effects of prescribed fire, or treatments including prescribed fire (e.g. thinning, or herbicide treatment in combination with Rx fire) in eastern North America.

Fire Management Objective	Ecosystem	Location (State/ Province)	Ecological Division	pH	P	K	Ca	Mg	Na	Citation
Fuel reduction	Longleaf pine	LA	230		-/=	=	=			Haywood 2007
	Pine barrens	NJ	230		+	+	+	+		Gray and Dighton 2006
	Prairie	AR	230	=	-	=	=	=	-	Brye 2006
Habitat	Longleaf pine	GA	230		=					Boring et al. 2004
	Outwash sandplain	MA	220	+/=		-	=	=/-	-	Neill et al. 2007
Restoration	Grassland	MD	230	+	=	=	=	=	=	Sherman et al. 2005
	Oak savanna	MO	220	+	+	+	+	=		Rhoades et al. 2004
	Oak/Hickory	OH	220		-					Huang & Boerner 2007
	Oak/Pine	MA	220	+		=	=	=		Neill et al. 2007
	Pine-bluestem	AR	M230	+		=	+	=		Leichty et al. 2005
	Oak/Hickory - grass barrens	KY	220	=	=/+	=	=	=		Rhoades et al. 2002
Site prep	White pine	NC	M220	+		+	+			Knoepp et al. 2004
	Black spruce	NF	210?	+	+/+	-/=	+/=	+		Scheuner et al. 2004

Notes: Ecological Division labels are consistent with those presented by Cleland et al. (2005) and in Chapter 1 of this volume. A '+' indicates a statistically significant increase in the measured variable, '=' indicates no statistically significant change in the variable, and '-' indicates a statistically significant decrease in the variable following prescribed fire or treatments including prescribed fire. When two symbols are presented for a given variable, the first symbol represents a shallower soil depth (typically O-horizon soil), and the second symbol represents a deeper soil depth.

Table 4. Summary of selected soil biological effects of prescribed fire, or treatments including prescribed fire (e.g. thinning, or herbicide treatment in combination with Rx fire) in eastern North America.

Fire Management Objective	Ecosystem	Location (State/ Province	Ecological Division	Enzyme activity[*]	Soil respiration[†]	Plant responses	Organism responses[†]	Reference
Fuel reduction	Oak/Hickory	OH		AcPh =; PhOx + in thinned, = in second site			Bacterial activity + in burn and burn with thin	Giai and Boerner 2007
	Longleaf pine	LA				Pine leaf had lower [N] (one site) and higher [P] (both sites)		Haywood 2007
	Longleaf pine	LA				Root variables generally not affected by fire; root length shorter with summer burn		Sword-Sayer and Haywood 2006
	Red pine	MI				Fine root production not affected by fire		Zeleznik and Dickmann 2004
	Loblolly/longleaf pine	GA				Root biomass not affected by fire	Microbial N fixation + in clay soil but – in sandy soil	Lajeunesse et al. 2006
	Loblolly pine plantation	SC		AcPh + after 4 years in thin+burn sites; PhOx + after 4 years in thin+burn sites; Chit - after 4 years in burn only sites				Boerner et al. 2006
	Outwash sandplain	MA			=			Neill et al. 2007
Habitat Restoration	Oak/Hickory – Grass	KY			-			Rhoades et al. 2002
	Oak/Hickory	OH		AcPh +; PhOx +; Chit +; α-Gluc =; L-Glut =				Boerner et al. 2005
	Oak/Hickory	OH				N release from decaying roots was slightly higher in burned sites initially, but no difference after one year; no effect of fire on live root [N]		Dress and Boerner 2003
	Oak/Hickory	OH					Oribatid mites were reduced by annual fire, but not in less freq. fires.	Dress and Boerner 2004
	Oak/Hickory	OH				Root [N] was reduced in one site with fire but was not different in the other. Root [P] was not affected by fire.		Huang and Boerner 2007
	Oak/Hickory	OH			=			McCarthy and Brown 2006
	Oak	IL					Total epigeic collembolan density was not affected by	Brand 2002

						fire, but species richness declined with annual fire	
	Oak/Pine	TN/GA		- short term, = thereafter			Hubbard et al. 2004
	Oak/Pine	KY				Litter dwelling arthropods were negatively affected by fire and did not recover after 2 yr. Ground dwelling arthropods were not affected by fire (grasshoppers were more abundant in burned plots)	Coleman and Rieske 2006
Site prep	Jack pine	ON				Microbial community diversity was decreased 5 yr following site prep burn in a clear cut relative to unburned clear cut at the whole plot level. No difference in microbial diversity at finer scales.	Staddon et al. 1998a
	Jack pine	ON	AcPh in burned plots not differenct from clear cut and unburned, but lower than untreated stands. ArSu lower in burned plots relative to others.				Staddon et al. 1998b
	Red pine	ON			Experimental removal of O-horizon soils resulted in better emergence of pine seedlings, but addition of ash diminished this response.		Herr and Duchesne 1996

Notes:
*For enzyme column AcPh = acid phosphatase; PhOx = phenol oxidase; Chit = chitinase; a-Gluc = a-glucosidae; L-Glut = L-glutaminase; and ArSu = aryl sulfatase. See text for details regarding ecological significance of these enzymes.

† For Soil respiration and Organism response columns, '+' indicates as statistically significant increase in the measured parameter, '=' indicates no statistically significant change in the measured parameter, and '-' indicates a statistically significant decrease in the measured parameter following prescribed fire or treatments including prescribed fire.

Table 5. Mechanical fuels treatment practices and their relative potential for soil impacts in the eastern U.S.

Practice	How it's used	Mechanism	Modifiers
In-woods mulching Mowing/bushogging Chopping/crushing	Precommercial thinning, reduction of ladder fuels, site preparation	Equipment traffic	# passes, soil type & conditions
Commercial bole harvest	Ladder fuel reduction, Stand development (thinning), Salvage/Sanitation cuts, Regeneration cuts	Equipment traffic, low-nutrient product removal	Degree of harvest, tree age, species, soil type & conditions
Intensive harvest	Same as above, plus: Understory fuel reduction, biofuel production	Equipment traffic+[1], high-nutrient product removal	Degree of harvest, tree age, species, season of harvest, soil type & conditions
Harvest residue removal	Prepare site for regeneration, esp. planting	Equipment traffic+[1], high-nutrient product removal, soil displacement	Degree & method of removal, soil type & conditions

[1] + refers to the generally greater number of passes with intensive harvest and site preparation as well as a reduced amount of debris upon which equipment can be driven, which increases the potential for physical property change