Cumulative Effects of Fuel Management on the Soils of Eastern U.S. Watersheds

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Fuel management treatments in the Eastern United States encompass diverse activities that 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, and burning slash), use of herbicides, and pre-commercial or early rotation thinning; these activities probably impact the most land area in the East. On public lands that are managed for natural resources, the fuel treatment strategies often are more varied and can include herbicide applications and thinning, 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 they can be acute and profound such as the direct soil-profile disrupting disturbances associated with site preparation and logging. Because the functions of forest soil arise through complex interactions among physical, chemical, and biological components, this chapter will address the effects of individual fuel treatment practices on each of these components (Burger 1994).

A wide range of different ecosystem types occupies the eastern landscapes of 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 that influence the long-term development of soil and ultimately determine what type of soil will be found in a given location (Jenny 1941). Soils in the Eastern United States fall into nearly every order, and are classified into hundreds of series (see chapter 3). Here we attempt to review the effects of fuel management practices (specifically prescribed fire and mechanical fuel treatments) on soils of eastern North America by collecting and synthesizing available soil-related data from as many different ecosystem types and soil types as possible. The reviewed material is therefore necessarily very broad in scope.

Prescribed Fire Effects on Eastern Soils

Prescribed fire is probably the most widely used treatment for fuel reduction in the ecologic divisions of the Eastern United States (Cleland and others 2007). 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 promote certain desirable plant and animal species. Furthermore, fire serves a crucial functional role in many (if not most) wildland ecosystems of the Eastern United States. This relationship is particularly well known in the Subtropical Division (230) pine-dominated (*Pinus* spp.) ecosystems of the Atlantic and southeastern Coastal Plains, and equally so in Prairie Division (250) tallgrass prairie ecosystems of the Midwest. Prescribed fires are also increasingly used for fuels management in the Warm Continental (210) and Hot Continental (220) Division pine forests of the Lake States, but less is known about their effects on ecosystem properties. Finally, although the role of fire in eastern hardwood forests, primarily in the Hot Continental and Hot Continental Mountains (M220) Divisions is less well known than for pine forests, much work has been performed in recent years to shed light on this important question.

Physical Effects of Prescribed Fire

The predominant physical effects of fire on forest soils (table 1) include heat transfer, development of hydrophobic conditions, higher soil temperature, increased 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; these diffuse into soils and then condense around soil particles, causing them to be water repellent. The degree to which this process occurs depends on fire temperature, residence time, and the characteristics of the organic matter (DeBano 2000). The development of hydrophobicity in eastern soils does not appear to be a substantially negative consequence of prescribed fires—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 (*Quercus* spp.) savanna ecosystems of Missouri, but this phenomenon has yet to be directly measured (Rhoades and others 2004). These authors suggested that destruction of aggregate structure might partially explain the slow recovery of plant communities observed in soils where large downed logs had "burned out" in a prescribed fire. Such aggregate destruction may be related to the observed changes in soil texture, as well as changes in water infiltration and water-holding capacity of the soils impacted by the intense "burn outs." In any event, 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.

Increased soil erosion has been observed in wildfire-impacted areas, but evidence for large soil losses from erosion in burned areas is limited. For example, in relatively steep slopes (35 to 45 percent) in the Southern Appalachian Mountains of the Hot Continental Mountains Division, Swift and others (1993) observed localized movements of soil in an area that had been burned in a prescribed fire, but they also reported 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 ≤66 percent consumed in the fires. Perhaps more important than soil erosion from the prescribed burn is erosion associated with fire control activities to prevent escape, and in particular the use of plowed fire lines (Van Lear and others 1985).

Chemical Effects of Prescribed Fire

Carbon

The pools of carbon that are likely to be affected by prescribed fire include plant roots, total soil organic carbon, microbial biomass carbon, and "black" carbon (charcoal and soot). All of these pools are more or less tightly related to one another, and fire-induced changes in one pool are likely to be associated with changes in others. The magnitude of fire effects on soil carbon pools largely depends upon the intensity and frequency of fires, soil type, and forest type (table 2).

Table 1. Physical effects of prescribed fire or treatments including prescribed fire (such as thinning or herbicide application) on soils in Eastern North America

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

n/a indicates that no data is available.
+ indicates a statistically significant increase in the measured variable.
= indicates no statistically significant change in the variable.
– indicates a statistically significant decrease in the variable.
[a] 210 is Warm Continental, 220 is Hot Continental, M220 is Hot Continental Mountains, 230 is Subtropical, M230 is Subtropical Mountains (Cleland and others 2007).
[b] Symbols are consistent with U.S. Department of Agriculture Natural Resources Conservation Service soil taxonomy (Oi is litter layer, Oa is fermentation layer, Oe is humus layer).

Table 2. Effects of prescribed fire or treatments including prescribed fire (such as thinning or herbicide application) on carbon (C), black carbon (BC), total nitrogen (N), carbon to nitrogen ratio (C:N), ammonium (NH_4), nitrate (NO_3), nitrogen mineralization (N_{min}), and dissolved organic nitrogen (DON) in the soils of Eastern North America

Fire management objective	Ecosystem	Location (State/ Province)	Ecological division[a]	C	BC	N	C:N	NH_4	NO_3	N_{min}	DON	Reference
Fuel reduction	Many	Many	220, M220, 230	=	n/a	n/a	n/a	n/a	n/a	n/a	n/a	Boerner and others 2008
	Oak savanna	MN	210	n/a	n/a	–	n/a	+	=	+	+	Dijkstra and others 2006
	Oak-hickory	OH	220	=	n/a	n/a	n/a	=	=	n/a	=	Giai and Boerner 2007
	Pine barrens	NJ	230	n/a	n/a	–	n/a	+	+	n/a	n/a	Gray and Dighton 2006
	Prairie	AR		+	n/a	+	+	n/a	n/a	+	n/a	Brye 2006
Habitat improvement	Scrub oak	FL	230	–/=	+	–/=	n/a	n/a	n/a	n/a	n/a	Alexis and others 2007
	Longleaf pine	GA	230	–	n/a	–	n/a	+	=	n/a	n/a	Boring and others 2004
	Outwash sandplain	MA	220	=	n/a	=	=	=	=	=	n/a	Neill and others 2007a
Restoration	Grassland	MD		=	n/a	n/a	n/a	n/a	n/a	n/a	n/a	Sherman and others 2005
	Oak-hickory	OH	220	+	n/a	n/a	n/a	n/a	n/a	=/–	n/a	Boerner and others 2005
	Oak savanna	MO	220	=	n/a	=	+	=	+	n/a	n/a	Rhoades and others 2004
	Oak-hickory	OH	220	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	Huang and Boerner 2007
	Oak-pine	MA	220	=	n/a	=	=	n/a	n/a	n/a	n/a	Neill and others 2007a
	Oak-pine	TN/GA	M220	–/=	n/a	–/=	n/a	=	=	n/a	n/a	Hubbard and others 2004
	Pine-bluestem	AR	M230	+	n/a	+	+	n/a	n/a	n/a	n/a	Leichty and others 2005
	Mesquite savanna	TX	310	=	+	n/a	n/a	n/a	n/a	n/a	n/a	Dai and others 2005
	Oak/Hickory - grass barrens	KY	220	n/a	n/a	n/a	n/a	+	=	+	n/a	Rhoades and others 2002
Site preparation	White pine	NC	M220	n/a	n/a	=	n/a	+	=/+	+/=	n/a	Knoepp and Swank 1997
	White pine	NC	M220	=	n/a	=	n/a	+	=/+	+/=	n/a	Knoepp and others 2004
	White Pine	NC	M220	–/=	n/a	–/=	n/a	n/a	n/a	n/a	n/a	Vose and Swank 1993
	Jack pine	ON	210		n/a	n/a	n/a	n/a	n/a	n/a	n/a	Staddon and others 1998
	Black spruce	NF	210	=	n/a	=	=	n/a	n/a	n/a	n/a	Scheuner and others 2004

n/a indicates that no data is available.
+ indicates a statistically significant increase in the measured variable.
= indicates no statistically significant change in the variable.
– indicates a statistically significant decrease in the variable.
Note: When two symbols are presented for a given variable, the first represents a shallower soil depth (typically organic horizon soil), and the second represents a deeper soil.
[a] 210 is Warm Continental, 220 is Hot Continental, M220 is Hot Continental Mountains, 230 is Subtropical, M230 is Subtropical Mountains, 310 is Tropical/Subtropical Steppe (Cleland and others 2007).

Plant root carbon

A large proportion of management-induced changes in soil organic-matter carbon can be traced to cumulative effects on carbon 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 Prairie Division tallgrass prairies of eastern Kansas, where fire frequency and the cover of warm-season perennial grasses are clearly related (Knapp and others 1998), and where more total root biomass can be found in frequently burned soils than unburned soils (Kitchen and others 2009). Because grass root tissues typically have very wide carbon:nitrogen ratios, the decomposition of this material is slower than analogous root tissue from forbs or woody species; the net effect is of larger accumulations of total soil organic carbon in systems that have higher warm-season grass cover (Knapp and others 1998).

Increases in perennial grass cover with frequent fire are also well known from forested systems such as the Subtropical Division longleaf pines (*P. palustris*) on the southern Coastal Plain (Brockway and Lewis 1997, Glitzenstein and others 2003) and loblolly and shortleaf pines (*P. taeda* and *P. echinata*) on the Southern Piedmont (Phillips and Waldrop 2008); and the Subtropical Mountains (M230) Division shortleaf-bluestem (*Andropogon* spp.) systems in the Arkansas Ouachita Mountains (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 true for the Ohio hardwood forests in the Hot Continental Division (Hutchinson and others 2005), and in jack pine (*P. banksiana)* systems in Ontario (similar to those found in the Warm Continental Division's Great Lakes States), where prescribed (site preparation) fires reduced grass cover in the first year following fire, but effects were negligible after the second year (Tellier and others 1995).

Soil organic carbon

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

Black carbon

Not all ecosystem carbon subjected to prescribed burning is volatilized to carbon dioxide. Depending on the fire severity, a fraction will remain in the ecosystem in the form of highly recalcitrant carbon (black carbon). The importance of black carbon in the total carbon cycle of fire-impacted ecosystems is increasingly being recognized (DeLuca and Aplet 2008). However, several aspects of the input and cycling of black carbon, for example in response to different fire frequencies, have not been thoroughly examined. Charcoal, elemental carbon, 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 black carbon 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 and others (2008) and Czimczik and Masiello (2007). Available published data on black carbon formation and its interactions are primarily derived from ecosystems with long fire-return intervals (DeLuca and Aplet 2008; Wardle and others 2008), and these systems likely will have black carbon dynamics very different from the pine savanna systems of the Southeastern United States. We have observed formation and storage of black carbon in the mineral soil horizons of a longleaf pine flatwoods site with an annual fire regime[1], and we expect this to significantly affect the net storage and turnover of carbon in these systems.

Nitrogen

Nitrogen 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 nitrogen concentrations in organic detritus (or necromass) are highly influential on the rate of detritus decomposition (Coleman and others 2004). Finally, the inorganic forms of nitrogen (nitrate or NO_3^-; and ammonium or NH_4^+), and the rate at which these forms are released from detritus or supplied by nitrogen-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 nitrogen cycling, particularly when fires are frequent. One of the principal effects is the volatilization and loss of nitrogen from the organic horizons of soil. This is directly related to the intensity of the fire and the relative proportion of the organic horizon that is consumed. Also important is the temperature at which combustion occurs and the depth to which high temperatures penetrate the organic horizon. 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 nitrogen volatilized. Temperatures <400 °C resulted in 90 to 100 percent loss of nitrogen whereas temperatures from 100 to 200 °C retained ≥75 percent of the original nitrogen content. The long-term consequences of nitrogen loss can be significant, whether through chronic loss from frequent repeated fires or through a large loss from a single high-severity fire. For example, a site that 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—an effect that was attributed to the loss of nitrogen capital from the system via volatilization (Elliott and others 2002).

In other aspects of nitrogen supply and cycling, however, prescribed fire has been demonstrated in many systems to have a positive effect. For example, nitrogen mineralization (microbial processing of organic nitrogen into plant available mineral forms) is either not affected by prescribed fire or is increased following prescribed fire (table 2). The net overall effect of prescribed fire on nitrogen 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 nitrogen, but fires of intermediate frequency or lower intensity may increase nitrogen availability.

Phosphorus

Phosphorus 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 phosphorus would be affected by fire occurrence (table 3), but the chemistry of phosphorus in soils is highly complex and usually is strongly influenced

[1] Callaham, M.A., R.J. DiCosty, and J.J. O'Brien [N.d.]. Unpublished data. On file with the Center for Disturbance Science, U.S. Department of Agriculture Forest Service, 320 Green Street, Athens, GA 30602.

by the pH (acidity or basicity) of soil. Because phosphorus is chemically bound to aluminum (Al) and iron (Fe) oxides at low pH, and similarly, is bound to calcium at higher pH (Schlesinger 1993), the availability of phosphorus in ash is somewhat dependent upon the pH of the underlying soil. Further complicating the chemistry of phosphorus in relation to fire is the fact that the ash produced by the fire has other constituents that 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 phosphorus 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 phosphorus becomes chemically bound with iron and aluminum (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 phosphorus availability. However, such effects are usually short term (on the order of months to a few 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 approximately), phosphorus can be volatilized and lost from ecosystems (Neary and others 1999), and as such, fire intensity can be of great importance to overall phosphorus availability following prescribed fire.

Other cations

In addition to the two macronutrients already discussed (nitrogen and phosphorus), several other essential nutrients may be affected by the incidence of prescribed fire in forested landscapes (table 3). The most widely studied of these are cations such as calcium, magnesium, potassium, and sodium. All these cations serve critical functions in various aspects of plant cell metabolism, and thus their availability for uptake can influence site productivity and even plant community composition to some extent. Because cations are typically not subject to volatilization, 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—plants, microbes, and animals all compete for them—the duration of fire-mediated spikes in availability is typically short and on the order of weeks to months.

Biological Effects of Prescribed Fire

Plant roots and fire

A large amount of information is available on responses of plants to fire in eastern forests (table 4). Effects range widely, from completely positive to completely negative, depending 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—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 a change in root distribution throughout the soil profile. In grasslands such as tallgrass prairie, annual fire causes roots to be distributed more deeply throughout the soil profile (Kitchen and others in press). In forested ecosystems, data on root distribution responses to fire 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 for the long term, fine roots proliferate in the organic horizons of the soil; but in frequently burned sites, the organic horizons are much reduced or eliminated completely, and thus fine root biomass is increased in mineral soil horizons (O'Brien and others 2010). The degree to which prescribed fire affects root distribution in other eastern ecosystems has not been extensively studied.

Table 3. Effects of prescribed fire or treatments including prescribed fire (such as thinning or herbicide application) on acidity or basicity (pH), phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg), and sodium (Na) in the soils of Eastern North America

Fire management objective	Ecosystem	Location (State/ Province)	Ecological division[a]	pH	P	K	Ca	Mg	Na	Reference
Fuel reduction	Longleaf pine	LA	230	n/a	–/=	=	=	n/a	n/a	Haywood 2007
	Pine barrens	NJ	230	n/a	+	+	+	+	n/a	Gray and Dighton 2006
	Prairie	AR	230	=	–	=	=	=	-	Brye 2006
Habitat improvement	Longleaf pine	GA	230	n/a	=	n/a	n/a	n/a	n/a	Boring and others 2004
	Outwash sandplain	MA	220	+/=	n/a	–	=	=/–	-	Neill and others 2007
Restoration	Grassland	MD	230	+	=	=	=	=	=	Sherman and others 2005
	Oak savanna	MO	220	+	+	+	+	=	n/a	Rhoades and others 2004
	Oak-hickory	OH	220	n/a	–	n/a	n/a	n/a	n/a	Huang and Boerner 2007
	Oak-pine	MA	220	+	n/a	=	=	=	n/a	Neill and others 2007a
	Pine-bluestem	AR	M230	+	n/a	=	+	=	n/a	Leichty and others 2005
	Oak-hickory - grass barrens	KY	220	=	=/+	=	=	=	n/a	Rhoades and others 2002
Site preparation	White pine	NC	M220	+	n/a	+	+	n/a	n/a	Knoepp and others 2004
	Black spruce	NF	210	+	+/+	–/=	+/=	+	n/a	Scheuner and others 2004

n/a indicates that no data is available.
+ indicates a statistically significant increase in the measured variable.
= indicates no statistically significant change in the variable.
– indicates a statistically significant decrease in the variable.
Note: When two symbols are presented for a given variable, the first represents a shallower soil depth (typically organic horizon soil), and the second represents a deeper soil.
[a] 210 is Warm Continental, 220 is Hot Continental, M220 is Hot Continental Mountains, 230 is Subtropical, M230 is Subtropical Mountains (Cleland and others 2007).

Table 4. Biological effects of prescribed fire or treatments including prescribed fire (such as thinning or herbicide application) on soils in eastern North America

Fire management objective	Ecosystem	Location (State/ Province	Ecological division[a]	Enzyme activity[b]	Soil respiration	Plant responses	Organism responses	Reference
Fuel reduction	Oak-hickory	OH	220	AcPh =; PhOx + in thinned, = in second site	n/a	n/a	Bacterial activity + in burn and thin/burn	Giai and Boerner 2007
	Longleaf pine	LA	230	n/a	n/a	Pine leaf had lower nitrogen at one site and higher phosphorus at both sites	n/a	Haywood 2007
	Longleaf pine	LA	230	n/a	n/a	Root variables were generally not affected by fire; root length was shorter with summer burn	n/a	Sword-Sayer and Haywood 2006
	Red pine	MI	210	n/a	n/a	Fine root production was not affected by fire	n/a	Zeleznik and Dickmann 2004
	Loblolly-longleaf pine	GA	230	n/a	n/a	Root biomass was not affected by fire	Microbial N fixation + in clay soil but – in sandy soil	Lajeunesse and others 2006
	Loblolly pine plantation	SC	230	AcPh + after 4 years in thin/ burn sites; PhOx + after 4 years in thin/ burn sites; Chit – after 4 years in burn only sites	n/a	n/a	n/a	Boerner and others 2006
Habitat improvement	Outwash sandplain	MA	220	n/a	=	n/a	n/a	Neill and others 2007a
Restoration	Oak-hickory-grass	KY	220	n/a	–	n/a	n/a	Rhoades and others 2002
	Oak-hickory	OH	220	AcPh +; PhOx +; Chit +; a-Gluc =; L-Glut =	n/a	n/a	n/a	Boerner and others 2005

continued

Table 4. Biological effects of prescribed fire or treatments including prescribed fire (such as thinning or herbicide application) on soils in eastern North America (continued)

Fire management objective	Ecosystem	Location (State/ Province	Ecological division[a]	Enzyme activity[b]	Soil respiration	Plant responses	Organism responses	Reference
Restoration (continued)	Oak-hickory	OH	220	n/a	n/a	Nitrogen release from decaying roots was slightly higher in burned sites initially, but remained unchanged after one year; fire had no effect on live root nitrogen	n/a	Dress and Boerner 2003
	Oak-hickory	OH	220	n/a	n/a	n/a	Oribatid – with annual fire, but = with less frequent fires	Dress and Boerner 2004
	Oak-hickory	OH	220	n/a	n/a	Root nitrogen was reduced in one site with fire but made no difference in the other; root phosphorus was not affected by fire	n/a	Huang and Boerner 2007
	Oak-hickory	OH	220	n/a	=	n/a	n/a	McCarthy and Brown 2006
	Oak	IL	220	n/a	n/a	n/a	Total epigeic springtails density = with fire, species richness – with annual fire	Brand 2002
	Oak-pine	TN/GA	220, 230	n/a	– short term, = thereafter	n/a	n/a	Hubbard and others 2004
	Oak-pine	KY	220	n/a	n/a	n/a	Litter dwelling arthropods were – with fire with no recovery after 2 years; ground dwelling arthropods = with fire; grasshoppers + in burned plots	Coleman and Rieske 2006

continued

Table 4. Biological effects of prescribed fire or treatments including prescribed fire (such as thinning or herbicide application) on soils in eastern North America (continued)

Fire management objective	Ecosystem	Location (State/ Province	Ecological division[a]	Enzyme activity[b]	Soil respiration	Plant responses	Organism responses	Reference
Site preparation	Jack pine	ON	210	n/a	n/a	n/a	Microbial community diversity – 5 years after burn in a clearcut compared to unburned clearcut at the whole plot level; microbial diversity = at finer scales.	Staddon and others 1998a
	Jack pine	ON	210	AcPh in burned plots not different from clearcut and unburned, but lower than untreated stands; ArSu lower in burned plots relative to others	n/a	n/a	n/a	Staddon and others 1998b
	Red pine	ON	210	n/a	n/a	Experimental removal of organic horizon soils resulted in better emergence of pine seedlings, but addition of ash diminished this response	n/a	Herr and Duchesne 1996

n/a indicates that no data is available.
+ indicates a statistically significant increase in the measured variable.
= indicates no statistically significant change in the variable.
– indicates a statistically significant decrease in the variable.
[a] 210 is Warm Continental, 220 is Hot Continental, M220 is Hot Continental Mountains, 230 is Subtropical, M230 is Subtropical Mountains (Cleland and others 2007).
[b] AcPh is acid phosphatase, PhOx is phenol oxidase, Chit is chitinase, a-Gluc is a-glucosidae, L-Glut is L-glutaminase, and ArSu is aryl sulfatase.

Soil microbes and fire

The effects of fire on soil microbes in eastern forests seems to depend to a large extent on the intensity of the fire. Joergensen and Hodges (1970) and Renbuss and others (1973) found that the responses of soil microbes to fires range from no detectable effect (low intensity prescribed fires) to total sterilization of the surface layers of soil (very hot wildfires). This early work focused primarily on the abundance of microorganisms and not their activity levels. However, others have observed that although there may be a decrease in abundance of microbes following fire, the remaining microbes can have higher activity levels than that of the preburn community (Poth and others 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 a year following fire. The nature and duration of microbial responses to fires in eastern forests are not well known. In one study examining soil carbon dioxide efflux (the combined production of carbon dioxide from plant root respiration and microbial and soil animal respiration) in loblolly pine stands of the South Carolina Piedmont, Callaham and others (2004) observed that soil respiration (one indicator of microbial activity) decreased in plots that had been burned or had been thinned and burned—a response attributed 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 the enzyme-based assay of microbial activity, which uses the actual concentrations of ecologically important enzymes in soils to make inferences about the makeup and function of the microbial community at the time of sampling; and the community carbon utilization profile, which uses an array of different carbon sources to evaluate the potential metabolic capacity of the microbial community from the sampled soils.

- The carbon utilization profiles give an estimate of microbial-community function diversity; 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.
- Changes in the concentrations of enzymes in soil can be attributed to changes in the relative importance of various functional groups of microbes. Of the many such enzymes present in soil, only a few are particularly well characterized and have standardized methods of measurement (Tabatabai 1982): 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 sulfur 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 (table 4).

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 springtails and mites) with increasing prescribed fire frequency in loblolly pine stands on the Coastal Plain of South Carolina. In this study, the authors compared the abundance of microarthropods in plots that had been burned every year, burned every 3 to 4 years, or left unburned for many years. They found that abundances of mites and springtails were reduced a small amount (~25 percent) by periodic prescribed fires, but that the reduction was dramatic (75 to 80 percent) with annual fires. Similar studies in midwestern Hot Continental Division forests showed similar results in that reduction of litter mass with prescribed fire generally reduced microarthropod numbers (Brand 2002, Dress and Boerner 2004). 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 Prairie Division grasslands such as the tallgrass prairie systems in eastern Kansas and Oklahoma; in these studies, microarthropods are decreased in abundance with frequent fire (Seastedt 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, much of which is lost in fires.

Macroinvertebrates and fire

The few scientists who have studied the responses of soil invertebrates to fire in forested ecosystems of the Eastern United States found that response is often driven by changes in habitat structure or by changes in the amount or the quality of food resources (Coleman and Rieske 2006). Thus, whenever fire affects vegetation, temperature, moisture, or the nutrient status of a soil, the potential exist 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). They found that the frequency of prescribed fires (plots burned annually, every 2 years, every 4 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 5-year study had negative responses to fire, but some groups were favored by fire. For example, among 28 different spider groups that were collected, four 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 a 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 populations and on the functions these beetles perform within the system.

Several studies on the responses of soil macroinvertebrates to fire have been conducted in Prairie Division grassland soils of eastern Kansas. 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 their abundance (James 1982). Interestingly, in areas close to human habitations (with nonnative earthworms present), prescribed fire had the effect of limiting the colonization into soils under frequently burned vegetation (Callaham and others 2003). Results of this study suggest that native earthworms in grassland soils are adapted to the warmer soil conditions often found under frequently burned vegetation; also that because fire improves the performance of grasses, native earthworms may have strong preferences for soils with abundant grass roots. This effect of fire on nonnative earthworms may have potential application as a control strategy in eastern forests where invasions of European or Asian earthworms are currently underway—this idea is in need of further research.

Mechanical Fuel Treatment Effects on Eastern Soils

Mechanical fuels treatments have the potential to alter soil properties and processes dramatically; but under many conditions they may have little to no impact on soils. These treatments affect soils by using heavy equipment, which may change physical and hydrological processes, and by cutting and removing vegetation and site organic matter (fuels), which changes soil fertility and soil chemical and biological processes (Powers and others 1990). Mechanical treatments can vary from single-entry understory mowing or mulching treatments with small tractors to multiple-entry whole-tree thinning and harvesting followed by harvest residue raking and piling (table 5). In addition,

Table 5. Mechanical fuels treatment practices and their relative potential for soil impacts in the Eastern United States

Practice	How used	Mechanism	Modifiers
Mulching, mowing, chopping, crushing	Precommercial thinning, reduction of ladder fuels, site preparation	Equipment traffic	Number of passes, soil type, and 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, and conditions
Intensive harvest	Same as above, plus: understory fuel reduction and biofuel production	Equipment traffic +[a], high-nutrient product removal	Degree of harvest, tree age, species, season of harvest, soil type, and conditions
Harvest residue removal	Prepare site for regeneration, esp. planting	Equipment traffic +[a], high-nutrient product removal, soil displacement	Degree and method of removal, soil type, and conditions

[a] + 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.

mechanical treatments are applied under stand and soil conditions that are both resistant and resilient to impact, or they can be applied in conditions that provide little resistance to soil disturbance or nutrient removal and few mechanisms for recovery. No mechanical treatment is without the potential for impacting soil function, but conditions do exist under which any mechanical treatment can be used effectively without degrading essential soil functions such as supply of 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 after soil disturbance if other practices, such as soil tillage and fertilization, are used (Fox 2000). These practices are feasible because they ameliorate damages and usually increase production. In extensive forest management systems practiced by families and other nonindustrial owners, especially those for whom timber yield is not the primary goal, the focus 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 mechanical treatments affect soil properties and processes is necessary to avoid degrading soil quality to the extent that natural processes cannot restore it.

Much of the basic knowledge we have regarding mechanical treatments 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. Unfortunately, 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. 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 on eastern forests.

Effects on Physical Properties and Processes

Mechanical treatments have the potential to cause changes to soil physical properties and processes (Greacen and Sands 1980, Lull 1959, Miwa and others 2004), and these changes have been linked to reductions in germination (Pomeroy 1949), establishment and early survival of seedlings (Bates and others 1993, Brais 2001, Foil and Ralston

1967, Hatchell and others 1970, Lockaby and Vidrine 1984, Scheerer and others 1994), sprouting or suckering success (Smidt and Blinn 2002, Stone 2002, Stone and Elioff 2000, Zenner and others 2007), seedling root growth (Jordan and others 2003, Mitchell and others 1982, Siegel-Issem and others 2005, Simmons and Ezell 1982, Simmons and Pope 1985, Tworkoski and others 1983), seedling shoot growth (Farrish and others 1995, Hatchell 1981, Lockaby and Vidrine 1984), 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 (Carter and others 2002, Reisinger and others 1993, Sanchez and others 2006, Scott and others 2007, Tiarks 1990, Xu and others 2000). Although the overwhelming majority of research on soil physical disturbance in eastern forests has been conducted in the pine forests of the Southern States or in the aspen forests of the North Central States, the general relationships hold for most forest types. Unfortunately, general relationships are often 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 harvesting, forest floor removal, and mineral soil disturbance; and all have evolved from those defined by Dyrness (1965) for Pacific northwestern forests. Miller and Sirois (1986) and Aust and others (1998) developed classification systems in the South, and Steber and others (2007) recently used a nationally based system to evaluate disturbance in the Great Lakes States. These disturbance classification systems are used widely for two reasons: first, they provide an easy and rapid assessment of forest sites; and second, unlike chemical or biological changes, soil physical disturbances have a clear and usually negative visual impact. Although visually based classification systems are useful for rapidly assessing and monitoring impacted areas, they are not generally effective at discerning quantitative changes in soil properties or processes (Aust and others 1998, Steber and others 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, resulting in an increase in bulk density 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 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 and others 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 (for example, resistance to penetration by roots), porosity and the distribution of pore sizes or quantity of air- or water-filled pores, hydraulic conductivity, and infiltration rate. Comparing bulk density among different soils is prone to imprecise interpretation because the bulk density 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 bulk density; and the more fine textured (clayey) a soil is, the lower its bulk density. Organic soils or topsoils with high organic matter content generally have the lowest bulk density. Within a given soil, comparing one bulk-density value to another is can also be misleading. A large absolute increase in bulk density from a relatively low value to a moderate value will have little effect on the properties that actually influence root growth—soil strength, aeration porosity, and water availability. Conversely, a small absolute increase in bulk density from an already elevated value to an even higher value will likely constitute soil damage. For example, an absolute increase in a loam bulk density from 1.2 to 1.4 mg/m^2 (0.2 mg/m^2 or 17 percent) is larger, both in absolute and relative terms, than an increase from 1.4 to 1.5 (0.1 or 7 percent). Under current U.S. Department of Agriculture Forest Service standards, a 17-percent increase in bulk density constitutes a significant impairment while the

7-percent increase does not, even though the increase from 1.4 to 1.5 would likely create much more growth-limiting conditions. Thus, change in bulk density is only useful given the initial or undisturbed value. For this reason, other parameters are better indicators of soil function.

The interactions among soil strength, aeration porosity, and water availability have been illustrated by Letey (1985) and have been updated by Da Silva and others (1994) with the creation of a single parameter, the least limiting water range. This parameter has been used successfully to explain loblolly pine response to soil physical disturbance (Kelting and others 2000), and although laborious and data intensive, could be used to monitor effects of soil physical disturbance on plant growth. Compaction increases soil strength, which becomes limiting to root growth at around 204 t/m^2 of pressure (Taylor and others 1966), although this value is species specific. Rutting and churning tend to decrease macroporosity and hydraulic conductivity substantially, and soils with <10 percent 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 influence on tree response, Aust and others (1998) suggested that soil strength is the best indicator of damage on dry to moist soils, the decrease in aeration porosity <10 percent 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, which have moderately well drained silt loam soils (Aquic Paleudalfs). One of the treatments 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 and others 2004). Planted pine biomass after five growing seasons was 5.9 mg/ha for no compacting, 7.2 mg/ha for moderate compacting, and 7.1 mg/ha for severe compacting (Stagg and Scott 2006). Competing understory biomass was 5.6, 2.0, and 1.8 mg/ha on the same plots, and these differences were statistically significant. Total biomass, however, was not significantly different among the compaction treatments. Furthermore, although most understory species were affected similarly, some species, such as flowering dogwood (*Cornus florida*) and some oaks were virtually eliminated from the compacted plots, presumably to the result of 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); whereas one plant may not respond negatively to a given change in soil properties or processes, others may be negatively impacted.

In rare circumstances, soil disturbance can create soil conditions that are actually more conducive to tree growth. If a site is characterized by coarse-textured or very loosely packed soils, water-holding capacity is often the soil property that influences tree growth. On these soil types, compaction can increase micropores by reducing the size of macropores; and even though overall aeration may decrease, water-holding capacity can be increased. This has been shown most definitively by Gomez and others (2002) in ponderosa pine (*P. ponderosa*) 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 should precede any management prescriptions that involve soil compaction.

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, mechanical treatments rarely cause erosion and sediment transport except on areas where the forest floor is removed, such as on main skid trails and roads. Although mechanical on treatments Eastern U.S. sites increased disturbance and water yield, measurable increases in sediment and nutrients are slight, especially where best management practices 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 impede drainage by reducing hydraulic capacity. Better drained soils more impaired by these treatments than inherently poorly drained soils (Aust and others 1995).

Effects on 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 direct removal of carbon and nutrients from the forested site. The factors that govern the cumulative removal of carbon 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 more 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 percent of the aboveground nutrients held in mature trees, and these materials represent an even greater percentage in smaller trees (Mann and others 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 compared to senescent leaves in the autumn, which lose 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 degrade soil function.

Concerns over harvesting and nutrient removal in eastern forests began in the early 1970s as a result of the work by 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 on nutrient-deficient Australian soils. Interest increased dramatically in the late 1970s during the energy crisis when whole-tree harvesting (clearcut harvesting of entire trees) was first being considered to provide biomass for energy. The result was a number of experiments across the Eastern United States that were designed to determine the potential nutrient loss from harvesting and other mechanical treatments.

The general nature of these nutrient loss experiments was regional because of differences in the management systems that were in place at the time. In the North Central States (Warm Continental Division) concerns generally focused on 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 Warm Continental Mountains (M210) Division of northeastern landscapes and in the less intensively managed southern forests in the Hot Continental Mountains Division, studies have focused on direct effects of whole-tree harvesting removals as well as the potential for increased leaching losses following the harvest. Finally, many of the northeastern studies also examined 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, a long term soil productivity program was installed in the 1990s in a variety of locations across southern and north central landscapes to examine both harvest intensity and forest floor removal.

Harvesting, especially whole-tree harvesting, removes large quantities of nutrients from a site (Freedman 1981, Kimmins and others 1985, Powers and others 2005). Recent reviews of long-term soil carbon and nitrogen responses to harvesting have shown little evidence that harvesting, even whole-tree harvesting, reduces soil carbon and nitrogen (Johnson and Curtis 2001, Johnson and others 2002, Knoepp and Swank 1997). 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 United States), and Johnson and others (2002) resampled five long-term studies in a variety of southeastern ecosystems. Evidence from long-term soil productivity studies (Powers and others 2005) indicate only slight decreases in soil carbon through 5 years since harvesting in Louisiana, no decreases in North Carolina (Laiho and others 2003), and no general decreases at 5 or 10 years post harvest across 21 installations (including the North Carolina, Louisiana, and Mississippi locations).

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

Harvesting-induced phosphorus removals have also been linked to reduced availability of phosphorus and growth declines. Yanai (1998) showed that whole-tree harvesting doubled the phosphorus removed compared to a similar bole-only harvested site and that harvesting reduced soil phosphorus net mineralization by 40 to 70 percent compared to an unharvested control. Scott and others (2004) compared whole-tree harvesting and whole-tree harvesting followed by forest floor removal to bole-only harvesting on Louisiana and Texas long-term soil productivity locations, and found that the former reduced extractable phosphorus compared to the latter by 23 percent; on North Carolina or Mississippi long-term sites, whole-tree harvesting and whole-tree harvesting followed by forest floor removal had no effect on extractable phosphorus. Scott and Dean (2006) and Scott and others (2007) linked loblolly pine productivity declines caused by whole-tree harvesting (compared to bole-only harvesting) to the preharvest quantity of extractable phosphorus in Louisiana, Mississippi, and Texas.

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

Effects on Biological Properties and Processes

Mechanical treatments affect soil biological functions both through physical disturbances to soil properties and processes and through impacts to organic matter and chemistry, but responses are quite variable. Because of this variability and complexity, few generalized statements can be made about the relationship between mechanical treatments and biological processes and properties.

Biological activity—commonly measured through carbon dioxide evolution, nitrogen mineralization, or enzyme assays—is usually more indirectly (than directly) affected by mechanical treatments. Biological activity depends on both substrate and aboveground environment, both of which are altered by mechanical treatments, as discussed above. Reducing forest cover warms soils, 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 mechanical treatments, activity decreases. These basic processes have been described in many ecosystems and forests and a detailed review is beyond the scope of this chapter.

In general, mechanical treatments have produced biological responses in places where the affected organisms specifically use the forest floor as habitat or are particularly sensitive to soil climatic conditions, such as reduced aeration and soil temperature. On the Missouri shortleaf pine-oak, long-term soil productivity sites, earthworm activity 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 ornate*, which is about 5 mm in diameter, while the density increased for Oligochaetes (*D. smithii*), which is about 2 mm in diameter (Jordan and others 1999).

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

Biological activity is clearly affected by soil disturbances caused by mechanical treatments, but responses are not consistent across treatments or soil types. Compaction reduced microbial biomass nitrogen in a Subtropical Division pine site (Li and others 2003), but changes in soil climate did not affect nitrogen mineralization. Neither compaction nor intensive harvesting affected soil carbon dioxide efflux on temperate hardwood sites in Missouri (Ponder 2005) 4 years after treatment, nor did intensive harvesting have an effect on a Subtropical Division pine site at 10 years after harvest (Butnor and others 2006). Although nitrogen mineralization was lower two and five years after compaction in North Carolina pine stands (Subtropical Division), harvest intensity had no effect on nitrogen mineralization; and the within-site differences in soil water content on the two soil types in the stands caused the greater differences in nitrogen mineralization than any treatment (Li and others 2003), similar to findings for microbial biomass and diversity discussed earlier.

Conclusions

One overarching conclusion that must be drawn from this review of soil responses to fuel management strategies in the Eastern United States is that the responses (chemical, physical, or biological) can be extremely context dependent. In other words, depending upon the conditions under which prescribed fires or mechanical fuel treatments are 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 damage to soils. Managers who take soils into special consideration when planning fuel management activities will minimize these intense perturbations. 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 fuel treatments likely to affect whole watersheds. Any models developed to assess whole watershed-level effects of fuel treatments on soils will likely be parameterized with the plot level data from the studies summarized in this review. Because such a modeling effort has yet to be undertaken, 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 and others 2004). These authors suggest that destruction of aggregate structure might partially explain observations of slow plant community recovery in soils where logs had "burned out" in a prescribed fire. Such aggregate destruction may be related to the changes in soil texture—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.

The response of roots to fire in eastern forests is an area needing 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 carbon 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 it moves into the soil profile (bioturbation or water infiltration), how it influences water quality, whether it enters the dissolved fraction of suspended organic carbon, whether microbial communities evolve to process it, and whether the its particle size 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 United States would benefit from a much more detailed accounting than is currently available.

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

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 (for example, nutrient mineralization rate versus loss or accrual of soil organic matter) will take different amounts of time to manifest themselves. 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. Scientists from the Forest Service and partner research organizations maintain long-term studies including soils-based studies, such as those on experimental forests and co-located long-term ecological research sites (established with National Science Foundation funding) as well as the long-term soil productivity plots described above. Such long-term experimentation will be critical to guiding the management of natural resources (including soil) in the future. The resulting data will be of great value when models are developed to fully address these issues at the landscape scale (Richter and others 2007).

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