Effects of Fire on the Ecology of the Forest Floor and Soil of Central Hardwood Forests

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Abstract

This paper reviews the scanty available database on the effects of dormant season fire on ecological processes and organisms in the forest floor and soils of forests of the central hardwoods region. Data from studies in other ecosystem types are used to supplement the existing database, but extrapolation across ecosystem types should be viewed with caution. Fires in the central hardwoods are typically low in intensity and consume primarily the unconsolidated leaf litter. As long as the fire can move across the open forest floor, soil temperatures generally do not increase enough to cause significant heating-induced mortality among soil-dwelling organisms. Soils under smoldering piles of woody fuels may, however, be subject to sterilization. Direct N volatilization is probably not an important pathway for fire-related nutrient loss due to low fire temperatures. The microclimate at the forest floor surface is probably affected significantly and this may produce phenological changes in root growth and microbial activity. More research in this area is warranted. Base cations released from dissolving ash may or may not increase soil pH and cation availability, depending on the nutrient status of the soil and the amount of ash deposited. Nitrogen availability typically increases after one or a small number of fires but may decrease over the long term. Abundances of soil animals in the forest floor are reduced by fire whereas those in the mineral soil are affected little. Recolonization of the redeveloping post-fire forest floor is rapid. Microbial abundances in the forest floor are typically reduced by fire, but rapid recolonization by these groups is also likely, except under smoldering piles of woody fuels. The database for the central hardwoods region is scanty and this area is greatly in need of additional research attention. Based on this scanty database, belowground impacts of low intensity, dormant season fires in the central hardwoods is likely to be less that the impact aboveground, though considerably more research covering a greater range of ecosystem types and fire frequencies is needed to verify this.

Introduction

The history of ecological studies indicating the importance of relatively frequent fire in the maintenance of forest composition and structure in eastern North America goes back over half a century. For example, Cottam (1949) concluded that frequent fire played a key role in the prehistoric ecology of oak-hickory *(Quercus-Carya)* forests in Wisconsin, and Daubenmire (1936) emphasized the role of fire in the vegetation of the “big woods” region of Minnesota. Based on pollen and charcoal profiles, Delcourt and Delcourt (1997) have determined that fire has been common in this region for at least 4000 yr. Their analysis of the fire record indicates, however, that the importance of local fires has increased and that of regional fires has decreased over time.

The importance of local, relatively small fires in this region is clear from the analysis of fire occurrence over the last century in oak forests in Vinton County, Ohio done by Sutherland (1997). In this region, the great majority of fires occurs either during the dormant season (69%) or in early spring (25%), with large, regional summer fires being rare (Sutherland 1997). The mean fire interval for the Vinton County sites was 3.6 yr, with locally extensive fires occurring on the average of 7.5 yr (Sutherland 1997). Based on these and other similar records of fire in the central hardwoods region (e.g. Guyette and Cutter 1991), this remainder of this review will focus on the effects of relatively low intensity, dormant season fires on the ecology of the hardwood forest floor and soil.

Forest fires may affect the belowground parts of the ecosystem through direct heating and by consuming and/or changing the characteristics of the surface fuels (Figure 1). Such alterations to the ground surface may then affect belowground organisms and ecological processes through direct volatilization of N and S, microclimate alteration, and deposition, dissolution, and convection of ash. In the paragraphs that follow I attempt to review each of these impact components individually, and conclude with a review on the biological and ecological consequences of these impact components taken together. One important caveat is necessary, however, before proceeding: the direct database for the central hardwoods appears to be sparse, at best. This is particularly apparent when comparing the data from this region to what is available for western conifer forests, shrublands, and prairie (reviews by Wells et al. 1979 and Neary et al. 1999). My approach is to summarize the data from the central hardwoods and supplement where required with studies from other temperate, humid ecosystems.

Fire Behavior and Direct Heat Effects

Low intensity dormant season fires rarely produce the large flame fronts and extreme temperatures of dormant season fires in chaparral or growing season fires in conifer forests (Boerner 1982). Franklin et al. (1997) recorded surface fire temperatures during two prescribed burns in forests near the Kentucky-Tennessee border. They reported average temperatures at the surface of the forest floor of 226C in an oak-maple forest and 190C in a mixed oak site. Similarly, Boerner et al. (2000b) reported mean fire temperatures of 210C and 157C in two early spring prescribed burns in oak-hickory stands in southern Ohio, and Blankeship and Arthur (1999a) reported forest floor surface temperature ranges of 316-398C and 205-315C in two prescribed fires in oak-pine forests in Kentucky.

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The range of variability in forest floor surface temperatures that can occur in this region is well illustrated by the studies of prescribed fires in northern Indiana forest stands conducted by Cole et al. (1992). Low creeping fires in oak forests in their sites often burned only leaf litter and produced little surface temperature rise when burning downhill; in contrast, peak temperatures 15 cm above the surface in other stands reached 300-400C with very high spatial variability (Cole et al. 1992).

Variations in fire intensity are related to many factors, including forest floor biomass/depth, slope position, aspect, and angle, and fire weather (Franklin et al. 1997; Boerner et al. 2000b). As an illustration of this, Boerner et al. (2000b) compared the behavior of prescribed fires done in two sites over three days in which the weather changed from warm and relatively dry to cool and moist. The fire conducted during the relatively drier, warmer day attained average temperatures at 10 cm above the forest floor of 210C, with a strong variation in maximum fire temperature related to slope position and soil moisture content. In contrast, the fire conducted during cooler, moister conditions only attained an average of 157C and there was no significant correlation between maximum fire temperature and slope position or soil moisture content (Boerner et al. 2000b). In the former fire, 2/3 of sample plots attained maximum surface temperatures >200C at least transiently, whereas in the latter fire only 1/6 of sample plots reached 200C (Boerner et al. 2000b).

Fires of this intensity typically burn only unconsolidated litter and fine woody fuels, leaving the humus and upper soil layers uncombusted. For example, in the fires described above (Boerner et al. 2000b); litter loss from a single fire ranged from <30% to >80%, depending on landscape position and fire temperature, but the underlying humus layer was unaffected. These results were similar to those obtained in a series of late autumn prescribed burns in oak-pine forests in southern New Jersey in which Boerner (1983) observed losses of 50-70% of the litter mass but <5% of humus mass, as well as an analysis of the impact of a single winter fire on oak-pine forest in Kentucky which demonstrated losses of ~30% of litter mass but no humus (Blankenship and Arthur 1999b). Studies of fires in northern Indiana oak forests and Georgia piedmont forests have also indicated that the effects of direct combustion are typically limited to the unconsolicated leaf litter (Cole et al. 1992; Bender and Cooper 1968).

Although the underlying, humified portion of the forest floor and the upper mineral soil layers are typically not combusted directly in such fires, transfer of heat to the unburned forest floor and soil could produce ecological consequences if severe enough. For example, soil enzymes will denature when temperatures reach 70C even transiently, and temperatures >70C for 10 min will kill fungi, protozoa and some bacteria (Lawrence 1956).

In general, however, surface fires moving across the open forest floor do not have strong potential for severe soil heating because only approximately 5% of heat released by fire is partitioned to soil, and mineral soil is a very poor conductor of heat (Raison 1979). Although there are few studies that report mineral soil temperatures during fire in the central hardwoods, there are studies from other regions that can serve as examples of what one might actually expect in our region. For example Saa et al. (1993) recorded mineral soil temps at 5 cm depth were <50cm in fires in pine forests and gorse (*Ulex europaea*) shrublands in Spain. Heyward (1938) recorded soil temperatures during a fire with 1m flame height in a North Carolina longleaf pine (*Pinus palustris)* forest and found temperatures at 2.5cm into mineral soil of 40C just after the flame front passed and only 25C 45 minutes after the fire had passed.

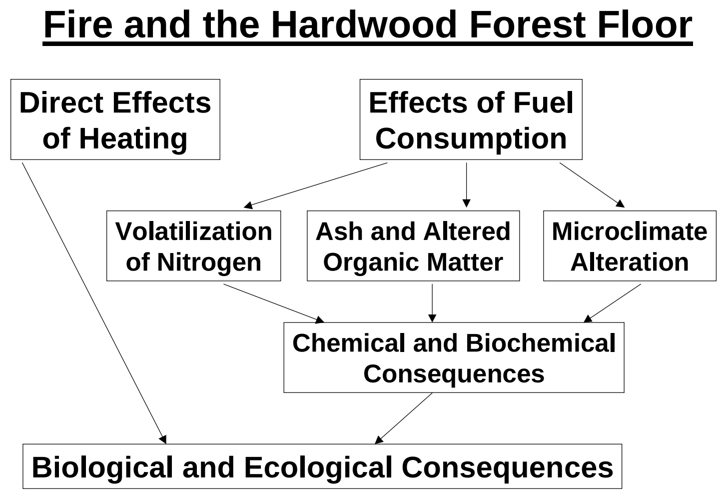

Figure 1.—Schematic approach for analyzing the direct and indirect effects of fire on organisms inhabiting the soil and forest floor.

Such fires can, however, have significant effects on temperatures within the forest floor layers that remain largely unconsumed by fire. Blankenship and Arthur (1999a) report temperatures as high as 315C in the upper 0.5 cm of the Oe layer during a fire in a Kentucky oak-pine forest that generated temperatures as high as 398C at the forest floor surface. Similarly, Cole et al. (1992) measured temperatures in the humus layer during prescribed fires and recorded ranges of 100-133C at 2 cm depth into the humus and 40-90C at 3 cm into the humus. Thus, low intensity fires moving across the open forest floor do not appear to have the potential to cause significant heating-induced mortality to organisms living in the lower humus layers or upper mineral soil, but may have such effects near the litter/humus interface.

The one situation in which direct, heating-induced mortality of soil organisms may become significant is when localized concentrations of woody debris continue to smolder in place for an extended period of time. Miller et al. (1955) monitored ground surface and soil temperatures during and after a fire in a New Zealand shrubland. The main fire area experienced maximum surface temperatures of about 200C, but there was no change in temperature recorded by sensors buried 5 cm and 10 cm into mineral soil. However, in areas where localized wood heaps burned for several hours, sensors recorded maximum temperatures of 100C at 5 cm and 60C at 10 cm depth in mineral soil. Thus, localized woody fuel accumulations do present a situation in which direct heating effects may have negative effects on soil biota.

Nitrogen Volatilization

Direct volatilization of N during combustion of litter can result in large losses of N from the ecosystem. In a review of the impacts of fire on nutrient cycling, Boerner (1982) reported net losses of 30-100% of litter N during fire as the result of direct volatilization and ash convection (Figure 2). However, direct volatilization of N does not begin until fire temperatures exceed 200C and does not become a major pathway for N loss until temperatures exceed 300C (Raison 1979; Boerner 1982). Given the ranges of surface temperatures recorded in fires in the central hardwoods noted earlier, it seems unlikely that direct N volatilization will be significant in this region. For example, Boerner et al. (2000b) estimated direct N volatilization from two prescribed burns in Ohio oak-hickory forests to be <1 kg/ha, less than 10% of annual additions to the site through atmospheric N deposition.

In addition to losses due to direct volatilization, nutrients may be lost from the forest floor via convection of ash (Boerner 1982). Monitoring of precipitation and ashfall in sites adjacent to active fires in western conifer forests (Clayton 1976) and southeastern pine forests (Lewis 1974) have demonstrated that nutrient redistribution via ash convection and deposition can have significant effects on nutrient budgets of neighboring sites.

Microclimate Alteration

Removal of the unconsolidated litter and deposition of blackened, partially combusted material on the ground surface have the potential to alter microclimate at the soil surface. Although such effects have commonly been reported in grassland ecosystems (Boerner 1982), to date I've been able to locate no published microclimate data for burned sites in the central hardwoods. However, in an unpublished study from southern Ohio, early spring surface soil temperatures in sites burned annually for three years were higher during the day and lower during the night than were those in soils of unburned controls (Personal communication. 1999. Louis Iverson and Todd Hutchinson, Northeastern Research Station, Delaware, OH 43015). Viro (1974) presented air and soil temperature data for burned spruce plantation in Finland. Soil temperatures were both higher in the burned site than under the canopy in a neighboring unburned site. However, soil temperatures in a canopy gap in the unburned site were intermediate between those of the burned site and those under the intact canopy. Thus, in this site at least, both the opening of the canopy and the alteration of the ground surface conditions contribute to microclimate alteration.

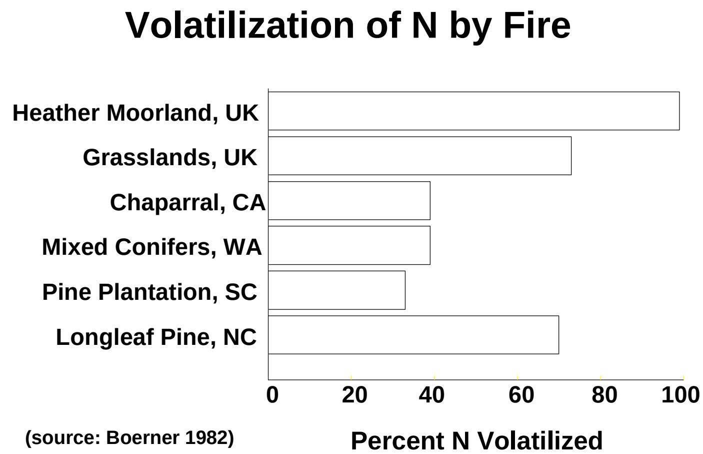

Figure 2.—Proporation of fuel N content lost to direct volatilization and ash convection. Original data sources are cited in Boerner (1982).

What impacts might earlier warming and more extreme conditions have on soil biota? In grasslands, the earlier warming of soils in burned sites contributes to an increase in the rate of N mineralization and nitrification, compared to unburned sites (Boerner 1982). This may well occur in central hardwoods sites as well but has not been explicitly measured. However, Dress and Boerner (submitted) observed that spring increases and summer decreases in fine root biomass in an Ohio oak-hickory site burned three times over three years were shifted 1-2 months earlier than in unburned controls or sites burned once two years before. The potential impact of ground surface microclimate alteration on belowground ecology in the central hardwoods remains largely unexplored.

Ash Deposition, Dissolution, and Their Effects on Soil Chemistry

The material that remains after fire has consumed part or all of the litter layer is a combination of partially-combusted organic matter and inorganic ash. This inorganic ash is rich in base cations such as Ca, Mg, and K, and is easily dissolved by rainfall. Whether the dissolution of this ash has subsequent effects upon surface soil chemistry depends, to a great extent, on the nutrient status of the soil prior to the fire and the mass of ash-derived nutrients that are added as a consequence of fire.

The impact of ash deposition and dissolution can be most vividly seen in studies of sites in which fire has been both frequent and spatially concentrated. Mikan and Abrams (1995) examined soil chemical properties in an oak forest site in southeastern Pennsylvania that had been used as the charcoal hearth for an iron plantation from 1771-1884. The soils under the former hearths were 0.25 units higher in pH, and base saturation increased from 10% in nonhearth soils to 38% in hearth soils. Furthermore, exchangeable Ca was 12.1X higher and Mg was 4.8X higher in hearth soils than nonhearth soils. Thus, the effects of repeated ash deposition could still be observed even 110 years after the last hearth fire.

A less extreme example comes from a study of the effects of burning of concentrated slash piles after logging in an East Anglia, England oak forest (Jalaluddin 1969). In this situation, the slash was concentrated in 1.5-2.0m diameter piles and burned, with active flames and smoldering lasting >3hr. Samples taken several days after the fires indicated that pH had increased from 6.0 in unburned to 9.0 under ash piles. However, within six months, soil pH under the former slash piles returned to preburn levels. Thus, the effects of fire on soil chemical properties are sensitive to both the amount of ash deposited in a fire and the number of fires that occur at that site over time.

Eivasi and Bryan (1996) have studied the impact of long-term prescribed fire on southeastern Missouri oak-hickory upland flatwoods. This experiment established in 1949, with annual burns and periodic burns (every four years) done in April prior to leaf-out. After 40+ years of burning there were no significant effects of burning on soil pH, Ca, K, or Al, and only a small indication of lowered Mg in annual burns. However, available P was only 24% of control in the annual plots and 35% of control in the periodic burn plots. Thor and Nichols (1973) report similar results from an annual and periodic burning study in the Highland Rim region of Franklin County, Tennessee. After 8 annual or 2 periodic burns, there were no statistically significant differences in pH among treatments. DeSelm et al. (1991) studied the same Tennessee site 27 years after the onset of burning. Again, the pH and availability of K and P of soils in the burned plots did not differ significantly from those of the controls. Thus, in the loess/limestone-derived soils of these two sites, lengthy periods of burning had no striking effect on base cations, but did impact P availability negatively in site with the longer history of burning.

Knighton (1977) studied the effects of 0, 1, 2, or 3 years of annual burning on an oak-hickory forest in central Wisconsin. Availability of Ca and Mg were considerably higher in the soil solution from the burned sites than the control, but with no indication of a cumulative effect over time. The soil solution concentrations of NO_3^- and PO_4^{3-} were also greater in the burned sites, and the average concentrations of both appeared to increase with the number of fires. However, Knighton (1977) points out that the increasing trend was not statistically supportable due to high among-sample variability in soil leachate samples. Blankenship and Arthur (1999b) report an increase of 0.2-0.3 pH units and a transitory increase in available N following a winter prescribed fire in an oak-pine stand in eastern Kentucky. They did not, however, report increases in extractable cations to accompany the increase in pH.

In our continuing study of annual and periodic prescribed fire in southern Ohio we have observed a somewhat different pattern of pH change. Our experimental design is based on four study sites, each of which is divided into three watershed-size treatment units. One of the units within each study area has been burned annually since 1996, a second was burned in 1996 and 1999, and the third remains as an unburned control. In 1995, prior to any burning, soil pH did not vary significantly among the three watershed-scale units in three of our four study areas (Figure 3A; Boerner, unpublished data). In the fourth site (Watch Rock), one unit had significantly greater pH than the other two, although we have still not yet determined why this difference existed. Following 2-4 fires, we now see a significant and positive effect on soil pH among all four study areas pooled and within two of the four individual sites (Figure 3B; Boerner, unpublished data). One site (Young's Branch) has not responded in soil pH to the burning. At Watch Rock, the one treatment unit with the unusually high pH has remained an outlier; however, the other burned unit at Watch Rock is now significantly greater in soil pH than its unburned control. Our study sites differ from those studied by Eivasi and Bryan (1996), Thor and Nichols (1973) and DeSelm et al. (1991) in that the soils of our sites were formed on nutrient and base poor sandstones and shales, not on relatively nutrient-rich loess and limestone as in the other studies. Thus, the impact of the base cations dissolved from the ash deposits is likely to be greater and more easily resolved in a nutrient-poor site such as ours than in the richer limestone and loess sites.

In a study of site preparation burning prior to restoration of Appalachian pine-mixed hardwood forests in North Carolina, Knoepp and Swank (1993) demonstrated that soil NH_4^+ concentrations and N mineralization rates increased after fire. The additional N made available from ash dissolution and from enhanced organic matter mineralization did not, however, result in increased stream water N concentrations; thus, the increased available N remained within the ecosystem.

Despite considerable combustion of surface organic matter by fire, few studies have noted changes in soil organic matter content. Knighton (1977) observed no effect of 1-3 fires on soil organic C in a Wisconsin oak-hickory site, and we have observed only slight increases in soil organic C in our Ohio oak-hickory sites (Boerner et al. 2000b); Boerner and Brinkman, submitted): Even after 40+ yr of burning in a Missouri oak-hickory forest organic C content in the mineral soil did not differ from that of unburned controls (Eivasi and Bryan 1996).

It should be emphasized again at this point that these patterns of effects are specific to dormant season, low intensity fires. The effects of high intensity wildfire may be quite different. For example, soil organic C and N availability were reduced by as much as half following intense wildfire in *Pinus halepensis-Quercus calliprinos* forests in Israel (Kutiel and Naveh 1987)

Biological and Ecological Effects

For convenience, I will review the biological and ecological effects of fire on soil organisms in decreasing size order: from roots to soil animals to microbes. The only study of the effect of fire on root biomass and production in the central hardwoods comes from our own study sites. Dress and

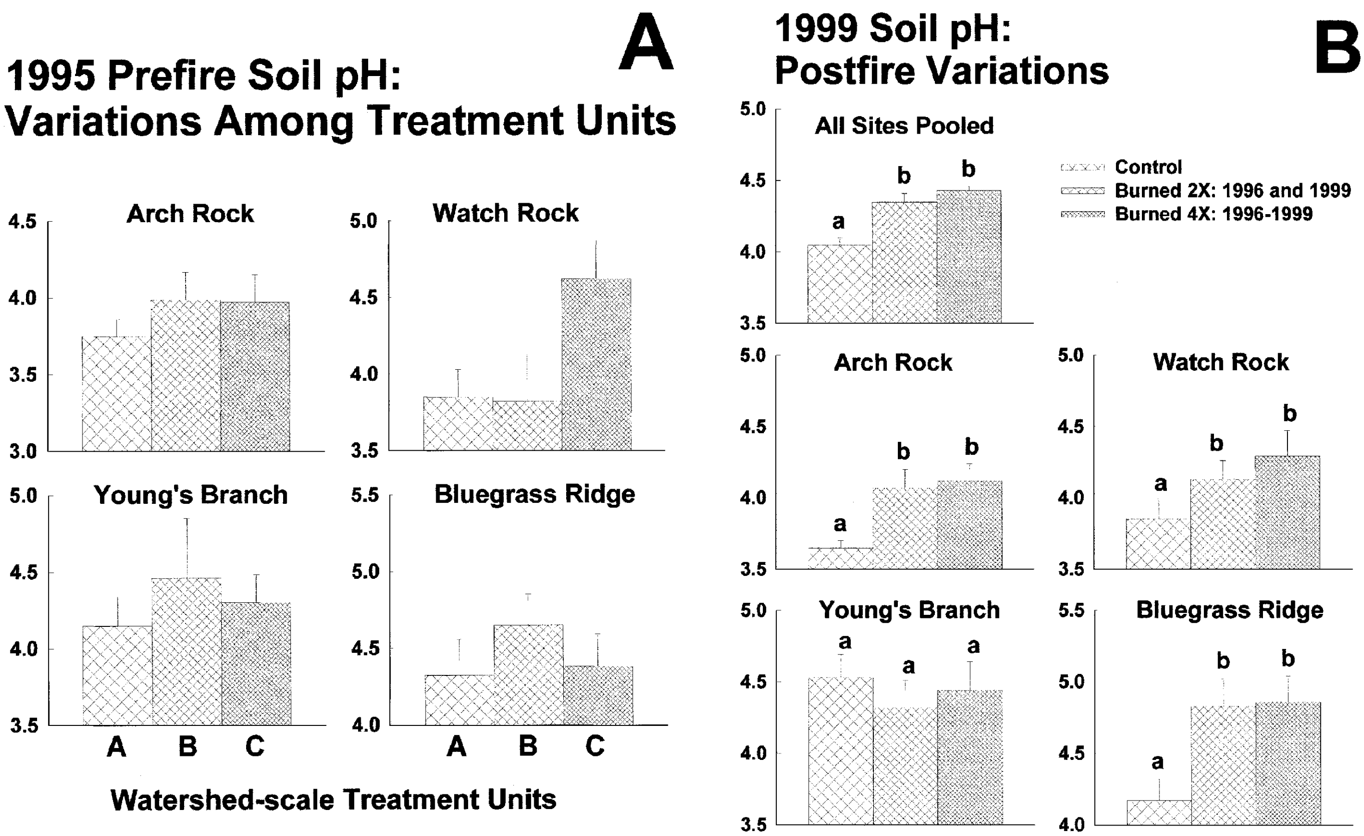

Figure 3.—A-horizon mineral soil pH in forested watersheds in southern Ohio. N=18 for each histogram bar (except for pooled data in B where N=72) with standard errors of the means shown. A. 1995 prefire variations among watershed-scale treatment units within the four study areas. B. 1999 postfire variations among treatment units. Histogram bars labelled with different lower case letters were significantly at $p<0.05$.

Boerner (in press) determined that both live and dead fine root biomass were significantly lower in a site burned annually for three years than in an unburned control. The variations in root biomass were inversely proportional to variations in soil N availability and N mineralization. This observation was consistent with both general models of the relationship between root biomass/production and N availability (Chapin 1980) and with studies in grasslands that also demonstrated reciprocal responses of roots and N availability to fire (Benning and Seastedt 1997).

The assemblage of small animals in the forest floor and soil of a forest ecosystems is both extremely diverse and critically important. These animals form a complex food web that is responsible for much of the processing of the detritus produced both above ground and below ground. Soil animals such as springtails (collembola), mites, and millipedes are important in breaking down coarse organic matter such as leaves into fine material that is suitable for microbial colonization and decay. Without soil animals present, leaf litter decomposition essentially ceases (Edwards and Heath 1963)

Once again, the database for the central hardwoods is sparse. In our study area, the abundance of oribatid mites in the top 15 cm of soil and forest floor decreases with increasing fire frequency (control, one burn, three annual burns), but the abundance of other mite groups was lowered only in the annual burn site (Dress, unpublished data). Metz and Farrier (1971) monitored the abundance of mites and collembola in the forest floor and soil of a North Carolina pine forest. In the forest floor layers, abundance of both groups decreased with increasing fire frequency. However, when fire was applied only every 4 years there was sufficient time for recolonization and population growth to near control levels. Abundances of collembola and mites in the mineral soil were not affected by either annual or periodic burns in these sites. Thus, on the basis of a scanty database we suggest that animals in the litter are likely to be more affected than those in the mineral soil and that effective repopulation can occur under a periodic but not annual burning regime.

The effects of fire on the soil microbial community can be evaluated by examining directly the effects on individual species or species groups or by determining the effect of fire on microbially-mediated processes such as N mineralization. Jorgensen and Hodges (1970) analyzed the impact of annual and periodic winter burns on the microbial community structure of a loblolly pine *(Pinus taeda*) forest on the South Carolina piedmont. In this study, the last periodic burn was eight years earlier and last annual burn was slightly less than one year earlier. In the mineral soil, none of the burning treatments significantly affected the abundance of either fungi or bacteria+actinomycetes, nor did burning affect the diversity of fungi in either the mineral soil or the forest floor. In the forest floor of the annual burn plots, however, the total abundance of fungi was only 78% of that in the control and the total number of bacteria+actinomycetes was only 18% of the control level. Thus, the microbial abundance in forest floor was strongly reduced by the annual burning. In contrast, microbial abundance in the periodic burn site was greater than in control, by 2.2X for fungi and by 1.2X for bacteria+actinomycetes. So recovery from burn happened in <8yrs and actually produced a forest floor environment more suited to microbial activity than the unburned control.

The potential of fungi present in the mineral soil to facilitate recolonization as the forest floor redevelops after fire is also supported by the work of Tresner et al. (1950) in hardwood forests of southern Wisconsin. They noted that although numbers of fungi decrease from forest floor down to mineral soil, and with depth in mineral soil, species composition does not change, so colonization of disturbed surface layers from deeper soil is possible without changing community structure (Tresner et al. 1950).

A more extreme example of the impact of fire on fungi comes from the work of Jalaluddin (1969) on the microbial ecology of soils under small plots on which concentrated piles of slash had burned and smoldered had continued >3hrs. At one week, three months, and six months post-fire, fungal abundance in the center of the former ash piles were only 3%, 6%, and 11% of that in areas >3m from fire piles. At the edge of the burned areas where mycelial recolonization would have supplemented colonization from new spores, one week, three month, and six month abundances were 17%, 31%, and 43% of those >3 m from the burned areas. Thus both mycelial ingrowth from areas surrounding the fire and new spore colonization occur. However, in this case, recolonization from below was prevented by the impact of the lengthy smoldering of the fire sterilizing the lower soil layers.

Mycorrhizal fungi are key symbionts for virtually all forest plants. There are two major groups of mycorrhizal fungi found in this region. The ectomycorrhizal fungi are a group of higher fungi (basidiomycetes and ascomycetes) which form symbioses with conifers, oaks, hickories, and beech. The arbuscular mycorrhizal fungi are a group of lower fungi (zygomycetes) that form symbioses with herbaceous plants and woody plants other than those that depend on ectomycorrhizae. In western conifer forests, some ectomycorrhizal fungi seem to be sensitive to fire while others are unaffected (Schoenberger and Perry 1982). Thus, in western conifer forests, how seedlings of a host tree species perform in recently burned sites may depend on which fungi that species depends upon to form ectomycorrhizae. Unfortunately, little is known about either the diversity of either group of mycorrhizal fungi in eastern forests or the response of these organisms to fire.

There may also be an interaction between effects on soil animals and effects on microbes. Lussenhop and Wicklow (1984) determined that fungal species diversity in Wisconsin prairie plots subjected to spring burning and raking in a Wisconsin prairie increased by 29% compared to the control. Moreover, the fungal propagules were also less aggregated and more evenly distributed in the burned plots. Lussenhop and Wicklow (1984) feel this change in spatial dispersion was due more to the increase in the abundance of mites and collembola on the burn plots than the actual burning and raking treatment because changes in fungal spatial distribution correlated most closely with changes in fauna.

Other than the study of Jorgenson and Hodges (1970) described above, studies of the direct impact of fire on bacterial abundances and/or community composition in the central hardwoods are uncommon. Blankenship and Arthur (1999b) did find a significant, positive effect of a single prescribed winter burn in an eastern Kentucky oak-pine stand on bacterial biomass and also reported a decrease in the fungal:bacterial biomass ratio following fire. Clearly the impact of fire on the forest floor and soil microbial assemblage continues to be neglected in fire research in this region.

Studies of functional measures of the impact of fire on microbes in the central hardwoods are more common than are direct community and density analyses. Vance and Henderson (1984) in a study of the same Missouri sites later used by Eivasi and Bryan (1996) found that N mineralization was reduced by long-term burning, but possibly not with just a single burn. As there was no change in soil organic C content due to fire, Vance and Henderson (1984) concluded that this long-term change was the result primarily of a change in the quality of the organic C in the soil. In contrast to what Vance and Henderson (1984) concluded, a progressive reduction in microbial activity or biomass is also a possible cause for this long-term reduction in N mineralization, and this is what Eivasi and Bryan (1996) reported 12 years later. Eivasi and Bryan (1996) demonstrated that microbial biomass was reduced relative to the control by 32% in the annual burn plots and by 21% in the periodic burn plots. In addition, Eivasi and Bryan (1996) demonstrated significant, burning-induced reductions in the activity rates of five key enzymes that serve as indicators of microbial activity: Acid phosphatase, a- and b-glucosidase, sulfatase, and urease. Vance and Henderson (1984) also report long-term reductions in tree growth on the burned plots, and attribute this to reduced N mineralization, thus linking the microbial effects to above ground ecological impacts.

In contrast to the Missouri case, studies done in many temperate ecosystem types have demonstrated an increase in N mineralization and N availability after a single or small number of fires (Boerner 1982). These increases are often attributed to the alteration of the organic matter by fire in such a way as to render it more susceptible to microbial decay, to increased microbial activity, and to altered microclimate. For example, Boerner et al. (2000b) demonstrated strong increases in N mineralization in plots burned thrice relative to those in those unburned plots. Plots burned once were intermediate in N mineralization rate. However, this observed increase in the activity of those fungi and bacterial involved in N mineralization probably cannot be generalized to all soil microbial groups. Both Saa et al. (1993) in pine forests and Boerner et al (2000a) in oak-hickory forests report decreased acid phosphatase activity following a single fire.

Summary and Conclusions

Fires in the central hardwoods are typically low in intensity and consume primarily the unconsolidated leaf litter. As long as the fire can move across the open forest floor, soil temperatures generally do not increase enough to cause significant heating-induced mortality of organisms dwelling in the mineral soil. Soils under smoldering piles of woody fuels may, however, be subject to sterilization. Direct N volatilization is probably not an important pathway for fire-related nutrient loss due to low fire temperatures. The microclimate at the forest floor surface is probably affected significantly and this may produce phenological changes in root growth and microbial activity. More research is in this area is warranted. Base cations released from dissolving ash may or may not increase soil pH and cation availability, depending on the nutrient status of the soil and the amount of ash deposited. Nitrogen availability typically increases after one or a small number of fires but may decrease over the long term. Abundances of soil animals in the forest floor are reduced by fire whereas those in the mineral soil are affected little. Recolonization of the redeveloping post-fire forest floor is rapid. Microbial abundances in the forest floor are typically reduced by fire, but rapid recolonization by these groups is also likely, except under smoldering piles of woody fuels. The database for the central hardwoods region is scanty and this area is greatly in need of additional research attention. Based on this scanty database, overall impacts of low intensity, dormant season fires in the central hardwoods is less that the impact on above ground organisms and ecological processes.

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