

COMMUNITY AND ECOSYSTEM RESPONSES FOLLOWING FIRE
IN THE LINVILLE GORGE WILDERNESS AREA

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A Thesis

by

SHAY DUMAS

Submitted to the Graduate School

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in partial fulfillment of the requirements for the degree of

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May 2004

Major Department: Biology

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ABSTRACT

COMMUNITY AND ECOSYSTEM RESPONSES FOLLOWING FIRE IN THE LINVILLE GORGE WILDERNESS AREA

(May 2004)

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Fire has been a natural occurrence in the southern Appalachians for millennia, as evidenced by the evolution and persistence of fire-dependent species. In the last century, fire suppression policies have resulted in changes in community structure that may be negatively affecting fire-dependent species, and we need to learn more about the interactions of fire and plant communities. In the fall of 2000, a wildfire burned much of the Linville Gorge Wilderness Area (LGWA), providing an opportunity to study the effects of fire in an area that had not been burned in 50 years. The objectives of my study were to assess changes in the biotic and abiotic environment caused by the 2000 fire.

Eight plots (30 m x 30 m) were established in nearby burned and unburned areas of the LGWA. Importance values were calculated for trees and shrubs in burned and unburned plots, and post-fire values were used to estimate pre-fire values in burned plots with the assumption that individuals dead aboveground but sprouting were alive prior to

the fire. There were few differences in pre-fire importance values for either the overstory or shrub layers between burned and unburned plots, and because the fire was mostly a low-intensity ground fire, there was no overstory mortality. However, the shrub layer, which was dominated by *Kalmia latifolia* (over 90% pre-fire importance value) was drastically affected. Almost all of the aboveground stems of *K. latifolia* were killed back by the fire, yet 86% of the shrubs survived and were sprouting two years after the fire.

Because of the loss of the *K. latifolia* canopy, light penetration (as assessed by hemispherical photography) was enhanced in the burned plots, which in turn, resulted in higher soil temperatures in the litter layer. Fire also significantly reduced the surface litter. These changes in the soil surface environment may have contributed to the higher plant species richness that was measured in the herbaceous layer of burned plots.

Paulownia tomentosa, an exotic invasive tree species, was found only in the burned plots.

Soil respiration rates, as measured with a Li-Cor 6200, were lower in the burned plots. Litter decomposition rates were significantly lower in burned plots after one year post-fire, most likely because higher soil temperatures resulted in drier litter. Both N and P accumulated at equal rates in the litter of burned and unburned plots.

The results of my study show that ground fires may contribute to the maintenance of fire-dependent species in the LGWA, but may also allow the introduction of exotic species. The changes in the microenvironment in the lower strata following the fire likely influence composition of the herbaceous layer as well as foster the re-accumulation of surface soil carbon via decreased litter decay.

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INTRODUCTION

Lightning fires have helped to shape the landscape of the southern Appalachians for millions of years, driving the evolution of fire-dependent species (Van Lear and Waldrop 1989). Ninety percent of lightning fires in the Appalachians occur between April and August, with the peak in May (Barden 1974). Typically, ignition occurs when lightning strikes a tall tree, or plants such as the ericaceous shrubs *Rhododendron* spp. and *Kalmia latifolia* which contain highly flammable compounds and encourage fire to climb back up into the canopy (Van Lear and Waldrop 1989). In order for fire to occur, there must be a lightning storm with little precipitation and the litter must be relatively dry. Lightning fires are generally not hot enough to cause stand-replacing crown fires in undisturbed habitats except on xeric, south-facing slopes. Consequently, it is on these sites where we tend to see fire-dependent species. Most fires are low-severity ground fires. However, crowning can occur if fire occurs during a substantial drought or where there are stands of pine killed or weakened by the pine beetle (Barden and Woods 1976).

Human impacts have led to substantial changes in the fire cycle of the southern Appalachians and have influenced community composition as well as management strategies. Fire frequency dramatically increased in the southern Appalachians upon the arrival of Native Americans 10,000 years ago (Van Lear and Waldrop 1989), who used fire to attract and surround game, to discourage catastrophic wildfires, and to increase soil fertility for agriculture and livestock (Whitney 1994; Brose et al. 2001). European settlers quickly caught on to the benefits of prescribed fire and continued using fire at

about the same frequency (Brose et al. 2001). Widespread logging in the southern Appalachians during the late 1800's and early 1900's left massive amounts of dried slash and set the stage for catastrophic wildfires (Brose et al. 2001). These fires were of much higher intensity and severity than the periodic fires used by early Americans, and they, along with other factors, led to a significant reduction in the magnitude of Appalachian montane coniferous forests (Brose et al. 2001). Another result of the uncontrolled fires during this time was the implementation of a fire suppression policy by the newly formed U.S. Forest Service.

Since the advent of fire suppression there has been a further reduction in the extent of xeric pine forests (Clinton et al. 1993; Brose et al 2001). Vose et al. (1999) suggested that pine communities were the most affected by fire exclusion, as they require fire for both maintenance and establishment. Many of the natural pine stands in the southern Appalachians were established following fire (Whittaker 1956; Barden and Woods 1976; Van Lear and Waldrop 1989; Frost 1998; Brose et al. 2001), and in the absence of disturbance, these pine stands, which are typically a sub-climax community, have succeeded to a community consisting of mixed hardwood stands and a dense heath understory (White 1987; Swift et al. 1993; Waterman et al. 1995; Elliott et al. 1999; Vose et al. 1999). These changes are marked by an increase in shade-tolerant hardwoods such as *Acer rubrum*, replacement of yellow pines by white pine (*Pinus strobus*), and a dramatic increase in the evergreen shrub *Kalmia latifolia* (Wade et al. 2000). *Kalmia latifolia* has been shown to inhibit growth and establishment of overstory species, due to its dense canopy and possible allelopathic properties (Chapman 1950; Whittaker 1956; Swift et al 1993; Waterman et al. 1995). This results in even-aged stagnant stands that

must compete for limited nutrients and resources.

One of the indirect effects of fire suppression is the increased frequency and severity in outbreaks of the southern pine beetle, *Dendroctonus frontalis* Zimmerman. Historically, fire and *D. frontalis* have interacted to maintain the structure and function of southern pine forests (Schowalter et al. 1981). In the absence of fire, pines face increased competition for light and resources and ultimately lose vigor (White 1987; Swift et al. 1993). Drought further weakens pines, and a severe drought in the 1960's contributed to a massive pine beetle infestation which led to the replacement of *P. rigida* by *K. latifolia*, *A. rubrum*, *Quercus coccinea*, and *Q. prinus* in the southern Appalachians (Whittaker 1956; White 1987; Clinton et al. 1993,1997). Stands of dead pine killed by pine beetle provide volatile fuels which further increase the probability of catastrophic fires (White 1987; Brose et al. 2001). Finally, catastrophic fires further dry soils, exacerbating drought conditions and promoting pine beetle infestation.

Since fire frequency has varied so much over time, it is difficult to establish what the natural fire regime is for the southern Appalachians. Several studies have attempted to quantify the desired fire regime to maintain healthy community function (Harmon 1984), which includes a high diversity in species composition and age classes, and a low incidence of invasive species, insect infestation, and disease.

The impacts of a fire depend on its severity, intensity, and frequency. Fire severity is assessed by impacts on soil properties, vegetation and water hydrology, and is a function of how much matter was consumed. Fire intensity, on the other hand, is a measure of how much heat was produced by the fire (Van Lear and Waldrop 1989). Fire frequency is the time over which the same area is burned again. The intensity and

severity of future fires often strongly depend on fire frequency. Low intensity fires tend to minimize erosion and nutrient losses and increase soil fertility (Van Lear and Waldrop 1989). These fires typically occur during spring when fuel loads are relatively moist, and often promote the growth of herbaceous plants and shade-intolerant oaks (Brose et al. 2001).

Fires of high intensity but low severity have been shown to benefit serotinous pines by reducing competition from overstory trees and the ericaceous shrub layer (Waldrop and Brose 1998; Van Lear 2000). Medium intensity fires have been shown to sufficiently reduce overstory competition and expose mineral soil without destroying seed banks (Welch and Waldrop 2001). High severity fires (meaning they reach the crown) typically result in complete mortality and ultimate stand replacement.

Fire seasonality is another factor determining subsequent recovery. Growing season fires have been shown to be most effective in removing a dense shrub layer and promoting pine and oak regeneration (Swift et al. 1993; Randles et al. 2002), whereas winter fires tend to have little effect on removal of competitors but do result in the partial removal of the litter layer.

Plants species vary in their susceptibility to fire. The ability of a particular plant species to survive a fire depends on such factors as water content, carbon and nutrient reserves, and phenological stage as well as fire adaptability traits (Debano et al. 1998). Plants are especially vulnerable to fires that occur during the growing season, and when flowering or forming seeds (White 1987; Debano et al. 1998). Plants with a high water content at the time of the fire face more of a threat from fire as heat is conducted more easily (Brose and Van Lear 1998). Temperatures above 64°C are instantly lethal to

protoplasm, though death can occur at lower temperatures if heat exposure is prolonged (Van Lear and Waldrop 1989).

Fire adaptive traits include thick bark, protected buds, basal sprouting, and serotinous seeds (Debano et al. 1998). Recolonization after a fire is promoted if that species has widely dispersed seeds (from nearby unburned sites), persistence of viable seeds in the ground or on trees, and vegetative sprouting (Debano et al. 1998).

The yellow pines of the southern Appalachians are highly adapted for existence in fire prone landscapes. *Pinus rigida* (Pitch pine) in particular, has many of the aforementioned traits. It has thick protective bark surrounding the bole, dormant buds at the base of the bole, as well as a basal crook in the root system. When the aboveground plant becomes stressed, sprouting occurs from the basal crook and dormant buds are stimulated to grow (Ledig and Little 1979).

Perhaps one of the most prominent adaptations to fire is cone serotiny. Cones of *P. rigida* and *P. pungens* (Table Mountain Pine) are sealed by a resin and seeds are released only when the coating is melted during the intense temperatures of a fire. If there is an abundant cone crop, these species will have a head start over the competition immediately following fire, and seedlings that establish during this time typically have the highest survival rates (Debano et al. 1998; Elliott et al. 1999). This is because fire typically removes the litter layer, leaving a nutrient rich bed for germination, while light and water are in abundant supply due to dieback of competitors. If conditions are not favorable for germination immediately after the fire, *P. rigida* seeds can remain viable in the soil for up to a year, and germination can be delayed until conditions improve (Ledig and Little 1979).

Another adaptation to fire and other disturbances is the relatively rapid maturity of serotinous pines. *Pinus rigida* forms viable seeds after three years, while its competitors (namely *Q. coccinea* and *Q. prinus*) do not form viable seeds until they are at least 20 years old (Little 1974). In areas where fire has been suppressed, some individuals of *P. rigida* and *P. pungens* have non-serotinous cones. Some individuals may have seeds that are released spontaneously, and others may have cones that open in response to an increase in ambient temperature (Barden 2000). These increased ambient temperatures have been shown to negatively affect seed viability, germination, and seedling survival in *P. pungens* (Barden 1977). While fire may not be needed to release seeds in all "serotinous" pines, light winter fires may be required to create favorable germination conditions (Little 1974).

Oaks also exhibit fire adaptive traits. Adults have relatively large root-shoot ratios and dormant buds that sprout following fire (Gilbert et al. 2003; Barnes and Van Lear 1998). Extensive root stocks produce high quality sprouts that have straight stems and are more resistant to root rot (Brose and Van Lear 1998). Sprouting is responsible for the bulk of the post-fire response in oaks, while seedling success is generally low because the acorns are typically damaged or consumed (Little 1979). Though acorns may be sparse, those that survive are positively affected by fire, through decreased insect predation and increased dispersal by birds and mammals (Brose et al. 2001).

Fire exclusion has led to the replacement of oaks by more shade-tolerant plants, except on very xeric sites (Abrams 1992). There is historical evidence that shows an increase in charcoal is correlated with a decrease in pine abundance and a subsequent increase in oak dominance, suggesting that increased fire frequency led to the

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replacement of pines by oaks (Abrams 1992). Pines are often able to persist in extremely xeric sites regardless of fire, due to their ability to cope with shallow and nutrient poor soils (Barden 1977). Upland pines tend to have their root systems in mineral soil, which doesn't experience the temperature extremes that the litter layer does during fire (Little 1979).

Ironically, traits that prove advantageous in frequently burned areas can be detrimental in areas where fire has been an infrequent, but recent occurrence. Despite having thicker bark, larger deciduous trees are often at an increased risk of death due to a greater buildup of flammable litter at the base (Debano et al. 1998). Cumming (1964) found greater post-fire mortality in overstory trees that had not experienced a recent burn compared to trees that had experienced more frequent fire. He found that in a previously unburned area, 97% of the oaks and 70% of the pines were killed by a wildfire, whereas the same age class experienced 42% and 0% mortality respectively in an area that had experienced light, but frequent fire. This difference in mortality was attributed to the high fuel loads that had accumulated in the previously unburned site. When fire has been excluded for long periods, plants may lose their adaptive traits to fire (Debano et al. 1998). Oak seedlings in infrequently burned areas may establish in thick litter layer and consequentially may have increased vulnerability to fire if roots are shallow.

The removal of the litter layer following fire impacts the biotic, physical, and chemical properties of the underlying soil. The rapid decomposition of organic matter by fire and subsequent leaching disperse nutrients more evenly into lower strata, thus increasing the potential distribution of fine roots (Aber and Melillo 1991). Another chemical and subsequent physical change occurs when the soil is dry and the fire is hot.

Hydrophobic organic compounds accumulate in the bottom of the organic layer and clog interstitial pore spaces forming a layer impermeable by water, resulting in further drying of the soils (Debano 1981). Dry soils can help root survival during fire by decreasing heat conduction. These changes in temperature and moisture in the soil microenvironment result in shifts in microbial activity.

In recent years, the importance of fire for improving habitat for flora and fauna, as well as in reducing future wildfires, has been reevaluated. Administrators of national parks such as Yellowstone, Grand Teton, and Sequoia-Kings Canyon have allowed lightning fires to burn under supervision in recent decades (Barden 1974). In the southeastern U.S., prescribed burning was advocated in the beginning of the 20th century, but stifled with the passage of the Clarke-McNary Act, which offered funding for fire suppression (Wade et al. 2000). Prescribed fire is now routinely used by land managers in the Piedmont and Coastal Plain. Fire suppression, however, has still been the prescription in the southern Appalachians because of fears that the extreme topography will lead to excessive erosion and unpredictable fire behavior (Van Lear and Waldrop 1989; Van Lear 2000). Nevertheless, many feel that since fire is a natural occurrence in the southern Appalachians, that fire suppression itself is the “disturbance” (White 1987; Wade et al. 2000).

During the fall of 2000 much of the southern Appalachians experienced extreme or severe drought conditions. On October 30, 2000 an unattended campfire escaped and burned approximately two-thirds of the nearly 5,000 ha Linville Gorge Wilderness Area (LGWA), located in northwestern North Carolina. The LGWA has a diverse assemblage of biota, which can be attributed to its deeply dissected landscape. The lower elevation

sites are occupied with mesic species such as hemlock (*Tsuga canadensis*) and rhododendron, and upper elevation sites contain more xeric oak and pine species. The last widespread surface fire in the LGWA occurred nearly 50 years ago, and subsequent fires have been suppressed (Newell and Peet 1998). In the past, lightning fires occurred at a high enough frequency to support fire-dependent species such as *P. rigida*, *P. pungens*, *Liatris helleri*, and *Hudsonia montana* on ridgetops and rock outcrops in the LGWA. Fire scar data indicate that the dry pine-dominated habitats historically experienced periodic low intensity ground fires every 5 to 7 years and catastrophic stand replacing fires every 75 years (Frost 1998). Despite the recent absence of fire, the 2000 fire was mainly a ground fire, although crowning did occur in stands of pine that had been killed by *D. frontalis*. The October 2000 fire, provided an opportunity to study an area that was shaped by fire, but which had not been subject to fire for nearly 50 years.

My objectives were to study the impact of the fire on the thermic-oak plant community in the LGWA, and on abiotic factors such as light availability and soil processes. I hypothesized that 1) a ground fire would mainly affect the shrub and herbaceous layers, 2) the loss of the shrub layer would increase the penetration of light to the forest floor, 3) the increased light would in turn increase soil temperatures and soil drying, and 4) warmer soils, coupled with recently released nutrients, would facilitate more rapid decomposition and nutrient cycling.

MATERIALS AND METHODS

Study Site

The study took place in the Linville Gorge Wilderness Area (LGWA), which comprises 4950 hectares on the eastern edge of the southern Appalachian Mountains in Burke County, North Carolina (Figure 1).

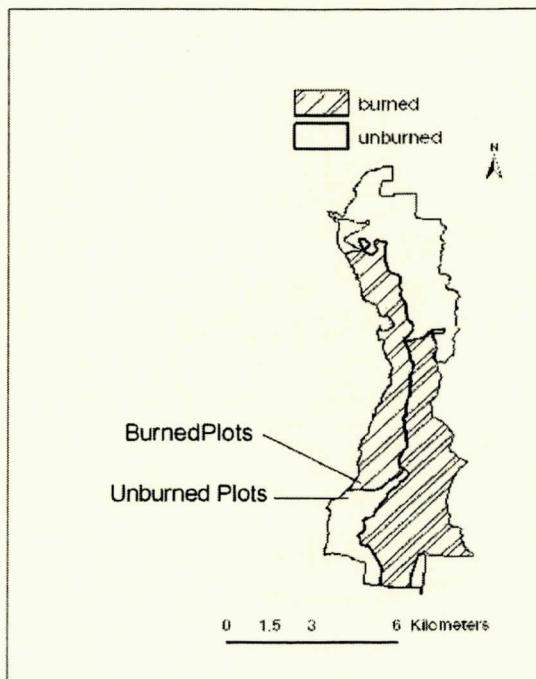


Figure 1. Map of Linville Gorge Wilderness Area showing areas burned in the November 2000 fire.

The study area consisted of two blocks, one burned and one unburned, separated from each other by approximately 1 km. Each block consisted of eight 30 m x 30 m (900 m²) plots, each with similar topological features such as altitude, aspect, and slope. Plots

were located using Geographical Information Systems and were in the thermic-oak-pine community type based on the work of Newell and Peet (1998) and Newell et al. (1999). I acknowledge that the blocks are pseudoreplicates, a constraint of the nature of the disturbance, yet I am confident that I have made every attempt to use a comparable reference site as my control (see van Mantgem et al. 2001).

Community Composition

Pre-Fire

Pre-fire vegetative composition was estimated by comparing measures of species richness and diversity of the canopy and understory between the burned and unburned plots. Pre-fire composition was estimated by assuming that in the burned plots, shrubs and trees that were standing-dead but sprouting from the root collar were alive prior to the fire. All trees over 30 cm diameter at breast height (dbh) were measured within each plot. The extent of the shrub layer and understory trees was recorded by measuring dbh of all stems between 10 and 30 cm within two 30 m x 5 m transects along the east and west margins of each plot. Total and relative basal areas were computed for each species in each plot. Density of each species (number of stems per plot) was recorded and relative density calculated as the density of a species in a plot divided by the total density of all species. If a tree or shrub had multiple trunks, it was counted as two individuals if the split occurred below 1.5 m. Frequency (number of plots in which each species occurred) and relative frequency (frequency of a species in a plot divided by the sum of all frequencies for a plot) were calculated. All data were expressed on a per hectare basis. An importance value (IV) was assigned to each species by summing the relative basal

area and relative density of each species and dividing by two. Values could range from 0 to 100.

Post-Fire

Density of standing dead trees and shrubs was similar between burned and unburned plots. Therefore, shrubs and trees that were marked as dead aboveground but were sprouting were considered to be a casualty of the fire. Since the sprouts were all under 10 cm circumference, they were not included in post-fire calculations of overstory IV. In order to assess the survival of *Kalmia latifolia*, which was the dominant shrub, a 5 m wide transect was run down the center for the length of each plot. All individuals within the transect were recorded as either alive, alive and sprouting, dead, or dead and sprouting. Although no attempts were made to identify standing dead vegetation, circumference at breast height was recorded.

Herbaceous Layer

All individuals \leq 1 m in height were considered members of the herbaceous layer (Gilliam and Roberts 2003), and were sampled for percent cover in mid summer 2002 using six 0.5 m by 0.5 m quadrats per plot. Notes were made of plant species in the herbaceous layer that were not captured by the quadrat survey.

Diversity Measures

Species richness was determined as the total number of species (trees, shrubs, and herbaceous plants) for each block. Species diversity was calculated using the Shannon-Weiner diversity index (Ludwig and Reynolds 1988). Evenness was determined using Pielou's index (Pielou 1975, 1977). Values were calculated for both pre- and post-fire estimates of community composition. Community similarity was computed for each

plant group (trees, shrubs, and herbs), as well as for all the plants using Jaccard's community similarity index (Mueller-Dombois and Ellenberg 1974).

Litter Mass

Total litter biomass of the surface organic horizon (surface organic horizon includes litter and organic layer) was collected in two 0.5 m by 0.5 m subplots in each plot prior to leaf fall on September 29, 2001. This material was air dried for 14 days to constant mass and weighed.

Hemispherical photography

The amount of light penetrating through the canopy and understory was assessed by using hemispherical photography. Photographs were taken at times of maximum leaf cover (September 20, 2001 and September 13, 2002), and minimum leaf cover (March 18, 2002), at two fixed points within each plot. A Nikon SLR camera was positioned horizontally at a fixed height of 93 cm using a self-leveling gimbal. An 8 mm Nikkor fisheye lens was used to provide a 180° view of the understory and canopy with the top of the camera facing due north. FujiChrome 400 color slide film was used, and slides were digitized in the Botany Department at the University of Georgia, Athens, GA and analyzed for Direct and Indirect Solar Flux using the CANOPY software program of Rich et al. (1998).

Soil Temperature

Soil temperatures were monitored using Thermochron ibutton temperature sensors (Dallas Semiconductors, Dallas TX). One sensor was placed in the humus layer in the

northwest corner of each plot. Sensors were periodically retrieved and data downloaded to a PC for analysis. Data were analyzed for daily means, maxima and minima. A Cumulative Heating Hour Index (CHH) was calculated as the cumulative temperature above 18.5°C for hourly mean temperatures greater than or equal to 18.5°C. This was done for each of the four seasons (spring, summer, fall and winter) and the index was standardized by dividing by the interval length. The same index was used to calculate a comparable Cumulative Cooling Hour Index (CHH), which was calculated similarly but for temperatures below 18.5°C.

Soil Respiration

Soil respiration was measured using a Licor 6200 affixed with the soil respiration chamber to measure CO₂ efflux (Li-Cor, Inc., Lincoln NE). Six soil collars were permanently located in the litter layer in four burned plots and four unburned plots. Respiration measurements began in November 2001 and continued approximately every six weeks until February 2003. Approximately 5 g of forest floor soil was collected most times when respiration was measured to determine moisture content. Soil was weighed immediately then dried at 60°C to a constant mass and reweighed to determine percent water content.

Litter Decomposition

Recently senesced litter was collected on November 22, 2000 from a thermic-oak-pine community in an unburned area near the plots. The litter was rinsed to remove ash that had deposited from the fires, then dried for 5 days at 60°C. Litter bags were made out of nylon screen with 1 mm x 1 mm mesh size. Blair et al. (1990) found that using single

species litter to assess natural decomposition rates led to an initial underestimate of N release and later to an overestimate of N accumulation, though no differences in mass loss were seen. For this reason, I used litter from multiple species in proportion to the abundances of the major species from the litter collected. Approximately 4 grams of red oak leaves, 3 grams of white oak leaves, and 2 grams of maple leaves were added to each bag. Eight bags were placed immediately inside the northwest corner of each plot on November 10, 2001. Two bags were retrieved from each plot on days 102, 194, and 289 after placement. After noting minimal difference in mass lost between replicate bags within a plot on the same date, I collected one bag from each plot on days 365 and 457. After collection each bag was dried at 60°C, re-weighed and the initial bag weight subtracted to determine remaining mass.

Nutrient Analyses

Litter from leaf bags was analyzed for nitrogen and phosphorus content at the North Carolina State Department of Soil Science Analytical Services Laboratory (Raleigh, NC). Samples were finely ground using a coffee grinder and an amalgamator (Darby Dental Westbury, NY) before being shipped for analysis. Litter from replicate bags in one plot was pooled for nutrient analysis. Samples were dry ashed, and diluted in HCl for phosphorus analysis on a Perkin-Elmer Optima 2000DV Ion Coupled Plasma Emission Spectrograph (Perkin-Elmer Corp., Norwalk, CT.). Nitrogen was determined using a Perkin-Elmer 2400 CHN Elemental Analyzer (Series II). At each sampling date the mass remaining of N and P was calculated by multiplying litter mass remaining by the concentration of N and P. Results were expressed as percent of original.

Statistical Analyses

Two sample t-tests were used to compare data from burned and unburned plots. A two-way analysis of variance was used for data collected over time, such as light penetration, soil temperature, decomposition, and nutrient fluxes . Data were checked to see that they met the assumptions for parametric tests. If not, they were transformed prior to analysis. Microsoft Excel was used to do the t-tests while the Statistical Analysis Software program (SAS Institute, Inc., Cary, NC) was used for the ANOVAs. Treatment differences were assumed significant when $p < 0.05$.

RESULTS

Community Composition

Pre-Fire

The pre-fire community compositions were similar for burned and unburned plots. The most abundant species in the overstory were *Q. coccinea*, *Oxydendron arboreum*, *Q. alba*, *P. strobus*, *Nyssa sylvatica*, *A. rubrum*, *Q. prinus*, and *P. rigida* (Figure 2a). No significant differences were seen in pre-fire composition between burned and unburned plots, with the exception of *A. rubrum*, which had higher values for density and basal area in the burned plots (Table 1). Mean basal area of all trees was 22.10 m²/ha for unburned plots and 23.37 m²/ha for burned plots.

Kalmia latifolia dominated the pre-fire shrub layer in both burned and unburned plots (Fig. 2a), comprising 87% of BA and 89% of density in burned plots and 89% of BA and 90% of density in unburned plots. Other species in the shrub layer included young individuals of the aforementioned canopy species, as well as *Rhododendron maximum* and in the unburned plots, *Symplocos tinctoria*. Mean total basal area of shrubs was 2.17 m²/ha and 2.42 m²/ha for unburned and burned plots respectively. While *K. latifolia* was by far the most abundant species in the shrub layer in all plots (Fig. 2b), there were differences in the composition of the remainder of the shrub community. *Nyssa sylvatica* had higher values for density and basal area in the unburned plots, while *A. rubrum* had higher values in the burned plots (Table 2).

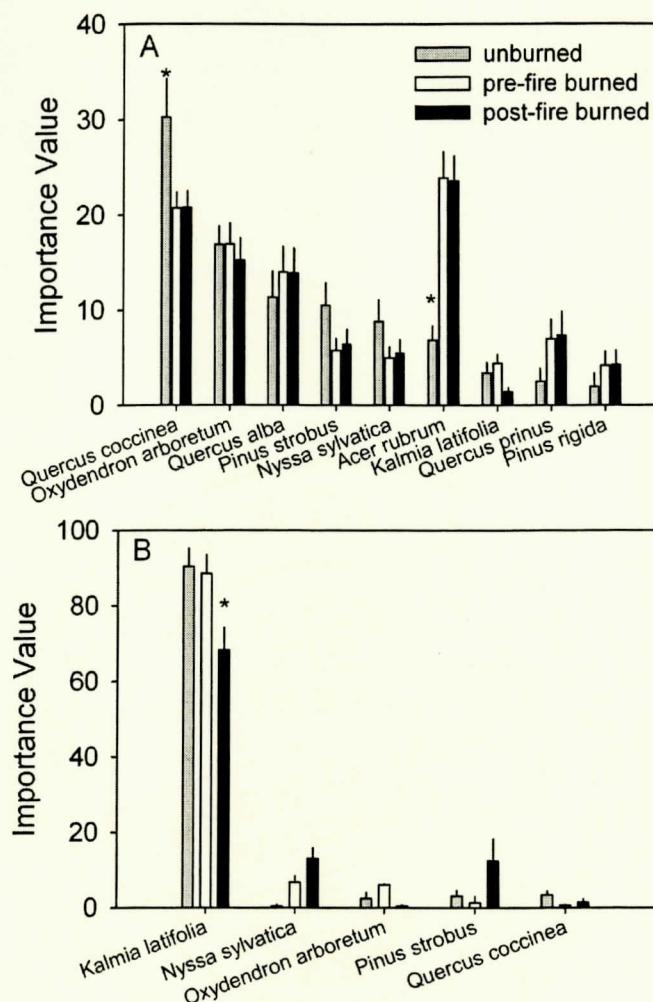


Figure 2. Importance values of (A) trees and (B) shrubs in burned (pre- and post-fire) and unburned plots. Values are means \pm SE; N = 8. Asterisks indicate significant differences ($p \leq 0.05$) between (A) unburned and pre-fire burned plots and (B) pre-fire and post-fire burned plots.

Table 1. Basal area and density of overstory trees (circumference ≥ 30 cm) in burned (pre- and post-fire) and unburned plots. Values are means \pm SE; N = 8. Bolded P values significant at $p < 0.05$

Species	Trees		Burned		Unburned		P values
	Basal Area (m ² /ha)	Density (stems/ha)	Basal Area (m ² /ha)	Density (stems/ha)	Basal Area (m ² /ha)	Density (stems/ha)	
<i>Acer rubrum</i>	0.45 \pm 0.064^a 0.41 \pm 0.041 ^b	190 \pm 34.2 178 \pm 33.3	0.18 \pm 0.043	39 \pm 8.9	0.005 0.003	0.698 0.797	0.078 0.635
<i>Quercus coccinea</i>	0.55 \pm 0.106 0.54 \pm 0.089	117 \pm 13.7 107 \pm 14.5	0.78 \pm 0.119	1699 \pm 23.4	0.173 0.904	0.150 0.953	0.055 0.639
<i>Quercus prinus</i>	0.16 \pm 0.053 0.16 \pm 0.019	36 \pm 8.1 31 \pm 8.3	0.06 \pm 0.035	14 \pm 6.8	0.464 0.985	0.464 0.933	0.654 0.933
<i>Quercus alba</i>	0.44 \pm 0.099 0.43 \pm 0.072	64 \pm 11.3 63 \pm 11.7	0.33 \pm 0.104	57 \pm 10.1	0.075 1.000	0.075 1.000	0.159 0.000
<i>Pinus strobus</i>	0.15 \pm 0.036 0.15 \pm 0.036	28 \pm 5.6 28 \pm 5.6	0.31 \pm 0.073	46 \pm 10.5	0.841 0.687	0.846 0.535	0.065 0.000
<i>Oxydendrum arboreum</i>	0.25 \pm 0.046 0.22 \pm 0.015	153 \pm 24.2 131 \pm 25.2	0.26 \pm 0.029	147 \pm 14.0	0.429 1.000	0.429 1.000	0.065 0.000
<i>Nyssa sylvatica</i>	0.14 \pm 0.027 0.14 \pm 0.027	23 \pm 6.1 23 \pm 6.1	0.18 \pm 0.043	76 \pm 23.8	0.122 0.664	0.122 0.642	0.137 0.000
<i>Kalmia latifolia</i>	0.01 \pm 0.003 0.01 \pm 0.000	17 \pm 4.2 14 \pm 4.1	0.03 \pm 0.011	38 \pm 12.0	0.146 1.000	0.146 1.000	0.804 0.000
<i>Pinus rigida</i>	0.16 \pm 0.066 0.16 \pm 0.066	15 \pm 7.3 15 \pm 7.3	0.05 \pm 0.038	13 \pm 8.2	0.258 1.000	0.258 1.000	0.239 0.000
<i>Rhododendron maximum</i>	-- --	-- --	0.01 \pm 0.014	100 \pm 9.7	0.239 1.000	0.239 1.000	0.000 0.000

^aPre-fire values. ^bPost-fire values.

Table 2. Basal area and density of shrubs (10 cm \geq circumference < 30 cm) in burned pre- and post-fire) and unburned plots. Values are means \pm SE; N = 8. Bolded P values significant at p < 0.05

Species	Shrubs		Burned		Unburned		P Values
	Basal Area (m ² /ha)	Density (stems/ha)	Basal Area (m ² /ha)	Density (stems/ha)	Basal Area (m ² /ha)	Density (stems/ha)	
<i>Kalmia latifolia</i>	6.312 \pm 0.977 ^a 2.257 \pm 0.660 ^b	2916.667 \pm 336.709 837.500 \pm 222.846	5.820 \pm 0.064	2675.000 \pm 387.440	0.720	0.642	0.002 0.004
<i>Acer rubrum</i>	0.581 \pm 0.121 0.455 \pm 0.127	145.833 \pm 36.153 95.833 \pm 36.967	0.039 \pm 0.028	12.5 \pm 8.768	0.478	0.007 0.350	
<i>Nyssa sylvatica</i>	0.028 \pm 0.018 0.028 \pm 0.018	12.500 \pm 8.768 12.500 \pm 8.768	0.255 \pm 0.073	83.333 \pm 23.570	1.000	0.016 0.020	
<i>Oxydendrum arboreum</i>	0.526 \pm 0.134 0.410 \pm 0.143	129.167 \pm 31.771 87.500 \pm 35.038	0.210 \pm 0.059	54.167 \pm 19.858	0.563	0.068 0.393	
<i>Pinus rigida</i>	--	--	0.026 \pm 0.018	8.333 \pm 5.455	0.199	0.170	
<i>Pinus strobus</i>	0.016 \pm 0.010 0.016 \pm 0.010	8.333 \pm 5.455 8.333 \pm 5.455	0.108 \pm 0.074	45.833 \pm 23.518	0.258 1.000	0.159 1.000	
<i>Quercus alba</i>	0.025 \pm 0.025 0.025 \pm 0.025	16.667 \pm 16.667 16.667 \pm 16.667	0.058 \pm 0.030	29.167 \pm 17.180	0.405 1.000	0.610 1.000	
<i>Quercus coccinea</i>	0.056 \pm 0.028 0.000 \pm 0.000	12.500 \pm 6.099 0.000 \pm 0.000	0.085 \pm 0.050	20.833 \pm 12.500	0.623 0.082	0.562 0.080	
<i>Quercus primus</i>	0.047 \pm 0.033 0.026 \pm 0.026	8.333 \pm 5.456 4.167 \pm 4.167	0.057 \pm 0.043	16.667 \pm 12.599	0.866 0.120	0.557 0.554	
<i>Rhododendron maximum</i>	0.096 \pm 0.096 0.096 \pm 0.096	25.000 \pm 25.000 25.000 \pm 25.000	0.009 \pm 0.009	8.333 \pm 8.333	0.398 1.000	0.543 1.000	
<i>Symplocos tinctoria</i>	--	--	0.023 \pm 0.015	20.833 \pm 13.998	0.170 1.000	0.180 1.000	

^aPre-fire values.

^bPost-fire values.

Tree species diversity using the Shannon-Weiner Index was found to be 0.80 for the burned plots and 0.85 for the unburned plots (Table 3). Jaccard's community similarity index was 0.90 for the pre-fire canopy layer of the burned and unburned plots. The diversity of the shrub layer was much less than that of the tree layer, mainly due to the domination of this layer by *K. latifolia*. Shrub diversity was 0.22 and 0.23 in the burned and unburned plots respectively (Table 3), while Jaccard's Coefficient of Community was 0.82.

Table 3. Species richness, diversity and evenness of burned (pre- and post-fire) and unburned plots. Values are means. N = 8.

Treatment	Species Richness	Species Diversity	Species Evenness
Trees			
Burned	9 ^a	0.80	0.84
Burned	9 ^b	0.80	0.84
Unburned	10	0.85	0.85
Shrubs			
Burned	9	0.22	0.23
Burned	8	0.38	0.42
Unburned	11	0.23	0.22

^aPre-fire values.

^bPost-fire values.

Post-Fire

No overstory tree species were lost due to the fire. While there were decreases in importance values for some tree species following the fire (Fig. 2 and Table 1), none of these differences were significant, and did not affect tree species diversity or evenness. The mean basal area for live trees only decreased 4% to 22.42 m²/ha.

The shrub layer was significantly impacted by the fire. Mean BA decreased 60% to 0.97 m²/ha post-fire. *Kalmia latifolia* decreased in importance from 90% pre-fire to 68% post-fire (Fig. 2b). The fire killed the aboveground stems of 78% percent of shrubs, and of these, 86% survived as basal sprouts. Species evenness increased from 0.23 to 0.42 in the shrub layer and diversity increased from 0.22 to 0.38 post-fire, which are a result of the decreased presence of *K. latifolia*.

Herbaceous Layer

The herbaceous layer in both the unburned and burned plots consisted of herbs, woody seedlings, and various mosses. The most frequent herbaceous plants in the unburned plots included *Vaccinium* spp., *Galax urceolata*, and *Smilax rotundifolia* (Table 4). Though not captured in the quadrat sampling, *Monotropa uniflora* and *Goodyera repens* were present only in the unburned plots. Like the unburned plots, the burned plots also had moderate cover of *G. urceolata*, *Vaccinium* spp. and *S. rotundifolia*. However, several species were found exclusively in the burned plots, such as *Erechtites hieracifolia*, *Erigeron anuus*, and *Eupatorium rugosum*. Individuals of *Paulownia tomentosa*, an invasive exotic tree species, were present only in the burned plots, though they were not captured in sampling.

Table 4. Percent cover and frequency of occurrence of herbaceous layer in burned and unburned plots. Values are means \pm SE; N = 8.

Species	Percent cover		Frequency (%)	
	Burned	Unburned	Burned	Unburned
<i>Vaccinium</i> spp.	0.91 \pm 0.66	2.16 \pm 1.85	50.0	37.5
<i>Galax urceolata</i>	3.10 \pm 2.17	0.83 \pm 0.83	37.5	12.5
<i>Acer rubrum</i>	0.39 \pm 0.12	0.60 \pm 0.17	75.0	87.5
<i>Smilax rotundifolia</i>	0.81 \pm 0.37	0.52 \pm 0.91	87.5	75.0
<i>Moss</i> spp.	0.18 \pm 0.18	0.52 \pm 0.35	12.5	25.0
<i>Pinus strobus</i>	0.04 \pm 0.04	0.35 \pm 0.24	12.5	37.5
<i>Gaultheria procumbens</i>	--	0.10 \pm 0.10	0	12.5
<i>Symplocos tinctoria</i>	--	0.10 \pm 0.08	0	12.5
<i>Kalmia latifolia</i>	0.00 \pm 0.00	0.06 \pm 0.06	0	12.5
<i>Quercus coccinea</i>	0.02 \pm 0.02	0.06 \pm 0.06	12.5	12.5
<i>Pinus rigida</i>	0.29 \pm 0.20	0.02 \pm 0.02	50.0	12.5
<i>Betula lenta</i>	0.02 \pm 0.20	--	12.5	0
<i>Chamalierium luteum</i>	0.06 \pm 0.06	--	12.5	0
<i>Erechtites hieracifolia</i>	0.52 \pm 0.32	--	37.5	0
<i>Erigeron anuus</i>	0.62 \pm 0.48	--	37.5	0
<i>Eupatorium rugosum</i>	1.64 \pm 1.48	--	37.5	0
<i>Liriodendron tulipifera</i>	0.02 \pm 0.02	--	1	0
<i>Oxydendron arboreum</i>	0.71 \pm 0.42	--	37.5	0
<i>Robinia pseudoacacia</i>	0.04 \pm 0.04	--	12.5	0

Forest Floor

Mean forest floor litter mass was significantly lower in burned plots (p=0.0066).

Mean litter mass (\pm SE) was 556 ± 70.4 g/m² in unburned plots, and 288 ± 36.0 g/m² in burned plots.

Hemispherical photography

In the first summer following the fire (September 2001), burned plots had greater Direct Solar Flux (DSF) than unburned plots when the canopy was fully leafed out ($p = 0.006$; Figures 3 and 4). In the following April, before the leaves had returned, there was no difference in DSF. The second summer following the fire (September 2002) showed the same pattern as the first ($p = 0.004$), with a slightly smaller difference in mean DSF, probably due to recovery of the canopy the second year.

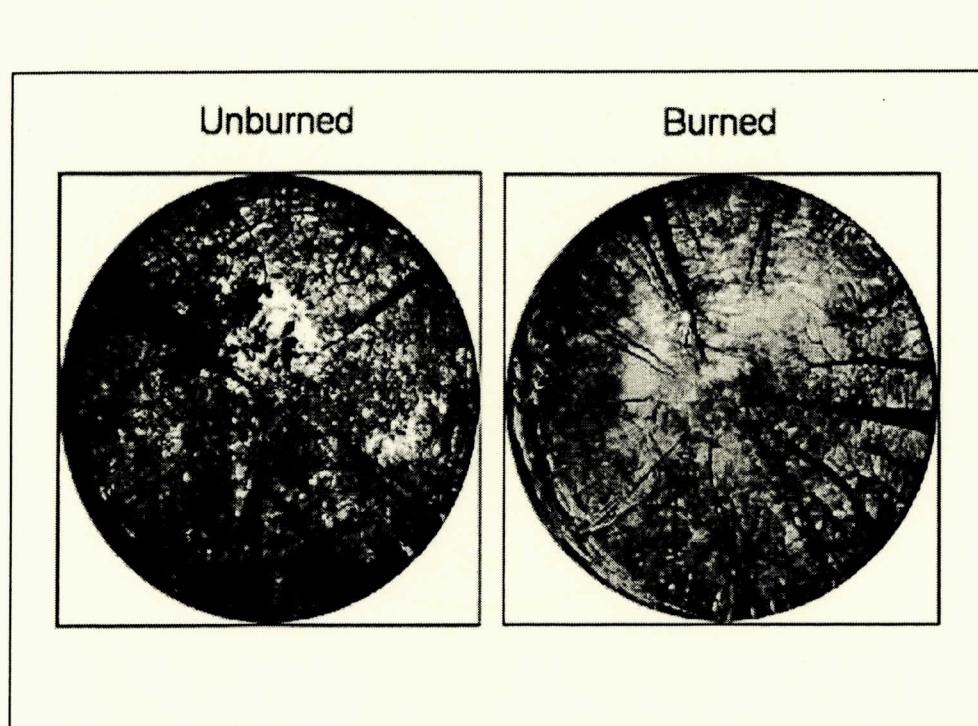


Figure 3. Representative hemispherical photographs of burned and unburned plots in September 2001.

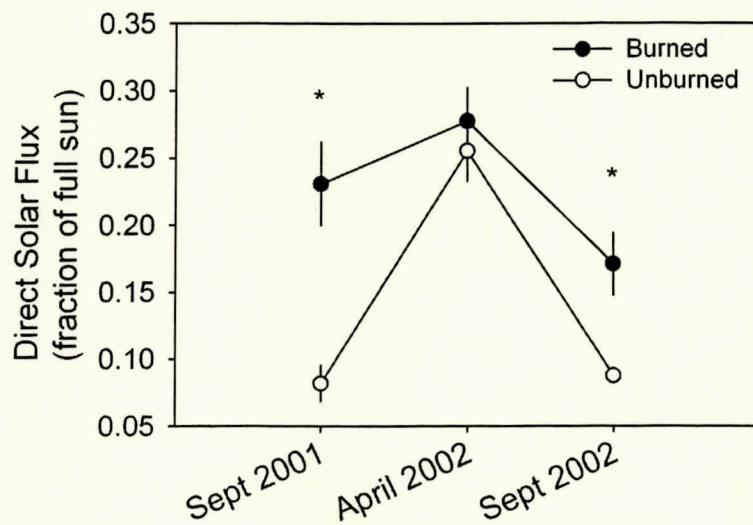


Figure 4. Direct Solar Flux in burned and unburned plots. Values are means \pm SE; N = 8. Asterisks indicate significant differences between burned and unburned plots ($p \leq 0.05$).

Soil Temperature

Burned plots had higher maximum soil temperatures during all seasons ($p < 0.035$), (Figs. 5,6). Mean temperatures were higher for burned plots during spring and summer ($p = 0.001$ and 0.003), and soil minima were higher in burned plots during summer ($p = 0.001$). The Cumulative Heating Hour Index (CHH) was significantly higher in burned plots during spring ($p = 0.003$), summer ($p = 0.003$), and fall ($p = 0.017$) of 2001 (Fig. 7). The Cumulative Cooling Hour Index (CCH) was higher in the unburned plots (Fig. 7) with significant differences in spring ($p = 0.001$) and summer of 2001 ($p = 0.001$).

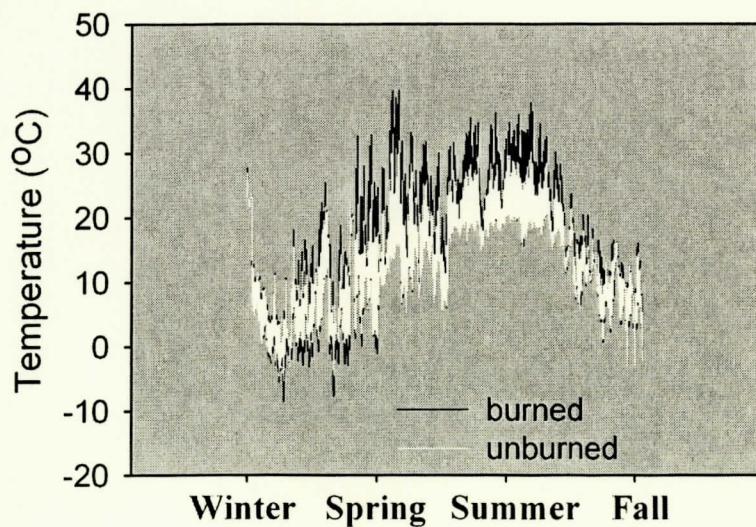


Figure 5. Surface soil temperatures for 2001-2002 in burned and unburned plots. Values are means of 8 plots.

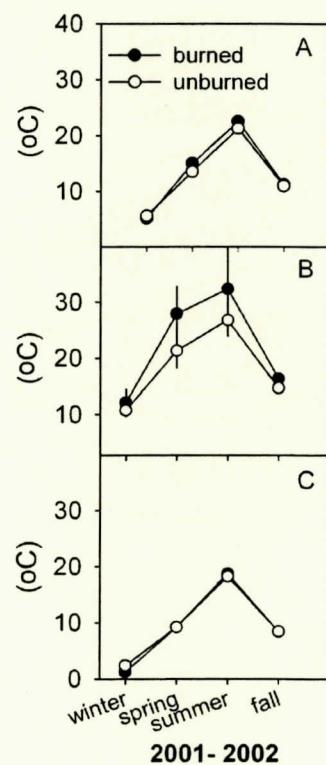


Figure 6. Mean (A), Maximum (B), and Minimum (C) soil temperatures for burned and unburned plots. Values are means \pm SE; N = 8. Seasons correspond to the following: winter (December-February), spring (March-May), summer (June-August), and fall (September-October).

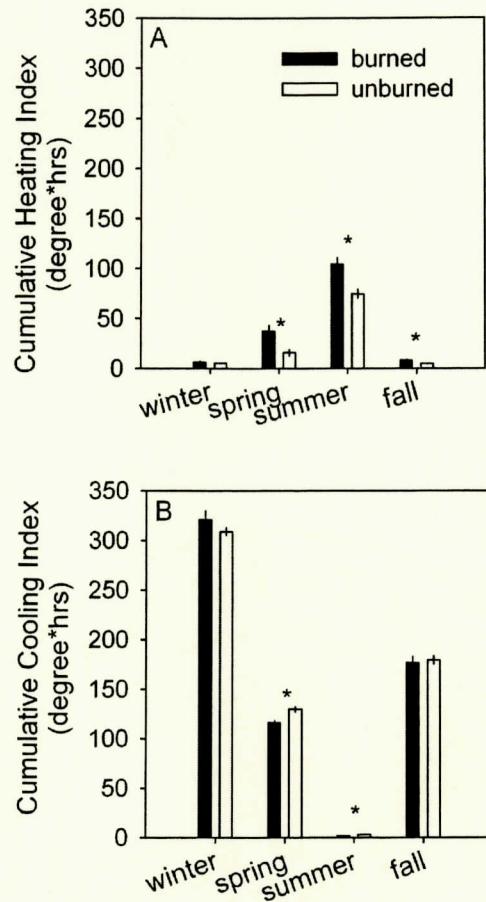


Figure 7. Cumulative Heating (A) and Cooling (B) Hour Indices for 2001-2002. See text for how to calculate indices. Values are means \pm SE; N = 8. Asterisks indicate significant difference between burned and unburned plots ($p \leq 0.05$).

Total Soil respiration

Total soil respiration (TSR) was consistently higher in unburned plots, with significant differences occurring during the summer and fall of 2002 (Fig. 8). In January TSR was lowest for both burned and unburned plots: mean (TSR) was 0.22 ± 0.031 g $\text{CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ for unburned plots and 0.14 ± 0.019 g $\text{CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ for burned plots. Rates were highest in July, the unburned plots having 1.26 ± 0.146 g $\text{CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ and burned plots 0.86 ± 0.141 g $\text{CO}_2 \text{ m}^{-2} \text{ h}^{-1}$.

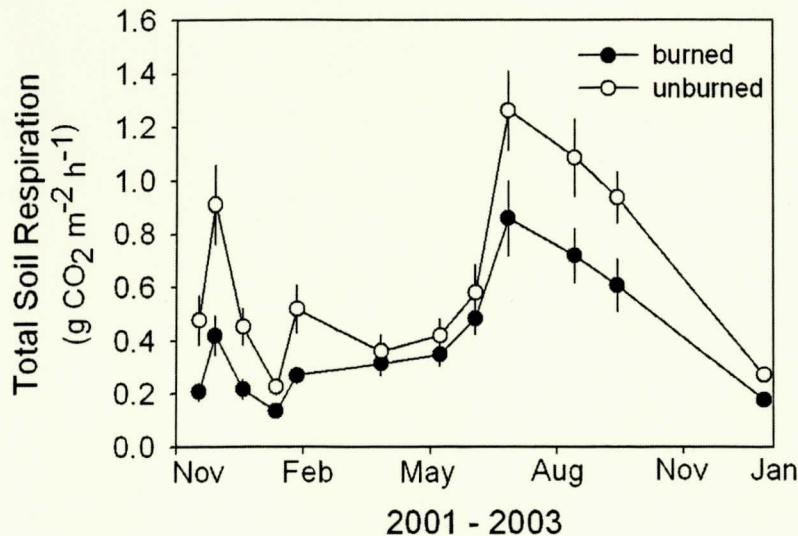


Figure 8. Soil respiration in burned and unburned plots for 2001-2003. Values are means \pm SE; N = 4.

Decomposition and Nutrient Analyses

Up to day 194, there were no differences in percent mass remaining between the burned and unburned plots (Fig. 9). After the litter bags had been out in the field for approximately nine months, the unburned plots began losing mass more rapidly than burned plots and this pattern continued until the end of the study ($p < 0.02$ for all remaining days). Percent of original nitrogen and phosphorus showed no differences between burned and unburned plots, with one exception. On day 102, phosphorus was significantly higher in unburned plots, but the lack of any other differences suggests this may be an aberration.

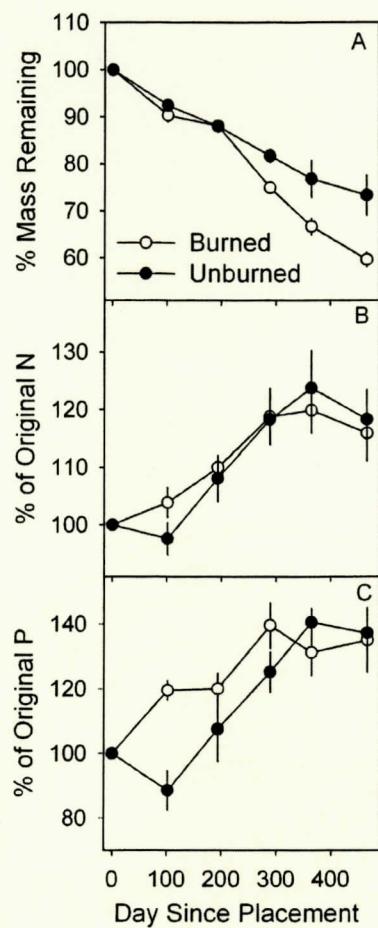


Figure 9. Percent of original litter mass (A), nitrogen (B), and phosphorus (C) in burned and unburned plots as a function of time since placement in the field. Values are means \pm SE; N = 8.

DISCUSSION

I investigated the interactions of biotic and abiotic factors following fire in a thermic-oak-pine community in the Linville Gorge. Fire affected the biotic environment primarily by removing the forest floor and herbaceous layer and by defoliating the shrub layer. Consequent abiotic changes included increases in light penetration and soil temperature, and decreases in soil moisture. These biotic and abiotic effects combined to increase understory diversity and cover, and to decrease litter decomposition and soil respiration. These results support my hypothesis that this ground fire would mainly impact the shrub and understory layers. The defoliation of the shrub layer resulted in increased insolation to the understory, which in combination with the removal of the litter layer led to increases in soil temperature and decreases in soil moisture. However, support was not found for the related hypothesis that increased soil temperatures would result in increased decomposition, nitrogen mineralization, and soil respiration. It is likely that the reduced soil moisture resulting from increased light penetration constrained later stages of litter decomposition and nutrient cycling, while the loss of the litter layer and resident soil fauna limited respiration.

The unburned plots were similar to the burned plots with regard to abiotic factors such as topography, slope, and aspect, as well biotic factors such as community composition, species richness and diversity, importance values, and total basal area. For example, pre-fire total mean basal areas and stem densities in the burned and unburned plots were within 2.6% and 5.8% of each other, respectively. With respect to pre-fire composition, the coefficient of similarity was 0.90 for the overstory and 0.82 for the

shrubs, and both sites were dominated by *Q. coccinea*, *Q. prinus*, *N. sylvatica*, *O. arboreum*, and *A. rubrum* in the overstory and *K. latifolia* in the understory. Given the similarities of the two locations prior to the fire, most of the differences in the burned and unburned plots resulted from the effects of the fire, and were not pre-existing conditions.

Fire frequency is one of the factors that determines the intensity and severity of subsequent fires and therefore, mediates the response of vegetation. In many studies, single fires have resulted in little or no impact on the overstory, a slight opening of the midstory, and an increase in density in the understory (Ducey et al. 1996, Welch et al. 2000) which may actually promote the growth of *K. latifolia* (Van Lear and Waldrop 1989, Arthur et al. 1998). This is consistent with my findings of prolific sprouting of *K. latifolia*. Infrequent fires may allow more fire sensitive species (namely *A. rubrum* and *N. sylvatica*) to outgrow fire susceptibility (Harmon 1984). This may be the case in my study, as none of the fire sensitive tree species were lost in the overstory.

The woody shrub, *K. latifolia*, suffered the most impact from the fire, whereas there was almost no effect on the canopy trees. The fire caused dieback of nearly 78% of the aboveground stems of *K. latifolia*, although 86% survived as basal sprouts. These results are similar to those by Elliott et al. (1999) who found that basal areas of *K. latifolia* decreased immediately following fire, but that ultimately densities increased due to prolific sprouting.

Dieback of *K. latifolia* resulted in increased solar radiation in the burned plots. While the fire initially reduced the cover of the herbaceous layer, the increased sunlight and loss of the litter layer resulted in increased diversity and abundance there within the first growing season. The distribution of plants in the herbaceous layer was patchy and therefore the statistics did not really capture the magnitude of establishment in the burned

plots. One of the most noticeable changes in the herbaceous layer was the rapid and plentiful recovery of the *Galax urceolata* following fire, which was due to rhizomes that survived where the organic layer was conserved.

Several of the species found only in the burned plots are known post-disturbance colonizers (*Erechtites hieracifolia*, *Erigeron annuus*, and *Eupatorium rugosum*) and their abundance may be attributed to seeds lying dormant in the forest floor that survived and flourished following the fire (Swift et al. 1993). Several individuals of the Asian invasive *Paulownia tomentosa*, which had not previously been recorded in the Linville Gorge, were present in the burned plots and rapidly increased in number and height during the duration of this study. Some individuals are approaching 4 m in height a little over 3 years post-fire. The persistence and proliferation of *P. tomentosa* in the LGWA may be an important issue for the communities there, as this plant is regarded as a noxious invasive in the southeast (Woods 1989).

Pinus rigida seedlings were more numerous and more frequent in the herbaceous layer of burned plots indicating that the fire promoted their establishment. Previous studies have shown that a fire frequency of 40-60 years will ensure *P. rigida* remains, and its importance has been shown to increase as fire frequency increases (Little 1974). Pine seedlings in particular are dependent on the removal of the litter layer in order to establish. Fire encourages growth and establishment by exposing mineral soil, increasing light availability, decreasing competition, and for serotinous pines - releasing seeds (Waldrop and Brose 1998).

The survival of the woody seedlings will most likely depend on their ability to outgrow the *K. latifolia* before it recovers and shades them. Clinton et al. (1993) found that 13 years after a fire there was no significant reduction in *K. latifolia*, although burned

plots had greater overstory diameter and lower density. The authors attributed this to establishment of shade-intolerant seedlings immediately following fire when *K. latifolia* was killed back, then subsequently less competition once the shrub layer recovered.

The prior abundance of *K. latifolia* has probably limited recruitment of oak and pine species in the LGWA, as evidenced by the low recruitment in the shrub and herbaceous layers of the unburned plots. Waterman et al. (1995) found that where *K. latifolia* was mechanically removed (thus increasing insolation) but the forest floor remained relatively intact, there were no changes in the short term successional dynamics of the community. These authors concluded that the remaining ericaceous litter inhibited seedling establishment, due to litter depth, allelopathy, or a combination of the two.

The increased light and the removal of the surface organic horizon also led to changes in the soil dynamics. Higher maximum soil temperatures in the burned plots may have contributed to an earlier growing season for plants in the herbaceous layer (Debano 1981), but may have also led to greater evaporation and increased drought stress later in the season. While total soil respiration is sensitive to changes in soil moisture to some extent, Fisk (unpublished data) found a lack of direct correspondence, suggesting that lower rates of respiration following fire were more likely dependent on the loss of the surface organic layer, rather than changes in soil moisture. This is probably due to the loss of soil microorganisms in the fire, as well as the loss of habitat for remaining populations (Reynolds and Hunter 2001).

While moderate increases in temperature can accelerate rates of decomposition, in this study the increased insolation, coupled with the loss of the surface organic horizon, further dried the already xeric soils and likely inhibited soil activity. In turn, this decrease in soil activity resulted in reduced litter decomposition rates in burned plots

during the growing season. The loss of the organic layer most likely reduced populations of soil microorganisms and remaining populations then had to cope with the changes in temperature and moisture (Knoepp and Swank 1993; Debano et al. 1998), thus limiting their activity.

The retention and release of N and P from decaying leaf litter is regulated primarily by physical factors, particularly during the early stages of decay, and by the nutrient requirements of decomposer organisms (Aber and Melillo 1980). Physical effects of fragmentation and leaching tend to impact litter nutrient content early in the decay process, and may be responsible for the early difference in litter N and P content in this study. Burned plots had slightly greater losses of both nutrients on the first collection date, during the first winter decay period. Differences in litter characteristics did not contribute to this pattern, as the same litter was used in both treatments; hence, this difference probably resulted from an effect of the fire on the abiotic environment. Two possible physical explanations are that: 1) greater temperature fluctuations in burned plots may have resulted in an increase of freeze and thaw events and consequent nutrient loss (Likens and Bormann 1995), and 2) the decreased cover in the understory decreased interception of precipitation and increased nutrient leaching. This effect was apparently overcome as temperatures increased with the onset of the growing season, and decomposer organisms immobilized N and P to the level required for decomposition, returning nutrient contents to similar levels in burned and unburned plots. It is interesting to note that immobilization of nutrients resulted in similar total quantities of N and P in litter, even while decay rates diverged and litter in burned plots lost more mass.

Forest floor litter (specifically the humus layer) provides the majority of nutrients available for plant uptake, and in southern Appalachian forests it can constitute about

50% of the available nitrogen (Vose and Swank 1993). In a fire, nitrogen can meet one of three fates: it can be volatilized, it can be deposited as ash, or remain in partially burned vegetation or detritus (Boerner 1982). The loss of N and its recovery are factors of what has burned, i.e. forest floor litter vs. woody debris. Fires of high intensity and low severity minimize impacts on the forest floor (Vose and Swank 1993). Forest floor nutrients are typically released during fire, while nutrients tend to remain in partially burned woody debris except in high intensity fires (Clinton et al. 1996). Nutrients that remain in ash on the soil surface may be transferred horizontally via wind or surface erosion or leached into lower soil strata following precipitation. Rarely is all organic matter combusted, as most fires are not hot enough or of long enough duration. This is evidenced by the presence of a layer of black partially organic material rather than a white inorganic layer (Boerner 1982). The steep slopes in the LGWA may have resulted in higher levels of erosion than in other studies.

Available nutrients may be retained on site through uptake by vegetation (Boerner 1982), and the increased light and temperature may have encouraged plant uptake of nutrients (DeBano et al. 1998). In a subsequent study conducted in my plots, Myhalyk (unpublished data) found that *K. latifolia* leaves from the burned plots had significantly higher N levels than leaves from unburned plots. This is supported by an earlier study (Clinton et al. 1996) that documented the role of *K. latifolia* in short-term nutrient retention following fire. This same study showed that forest floor nitrogen levels returned to 90% of original levels two years post-fire. Gilbert et al. (2003) found that seedlings of *A. rubrum* and *Q. coccinea* in burned plots had higher foliar concentrations of N, P, K, and Mg than seedlings in unburned plots. Early successional plants, with their high rates of growth and nutrient uptake also play an important role in the retention of

nutrients on site that would otherwise be lost (Boring et al. 1981).

However, these increases in foliar quality may make leaves more susceptible to herbivory in burned plots. Myhalyk (unpublished data) found higher levels of insect herbivory in leaves of *K. latifolia* following the 2000 fire. Other studies support these findings: Radho-Toly et al. (2001) attribute the post-fire nutrient pulse as the cause of elevated foliar nutrients which in turn results in increased herbivory.

Conclusions

In conclusion, the November 2000 fire in the Linville Gorge Wilderness Area had little impact on the overstory in my plots. Defoliation of the shrub layer resulted in increased light penetration to the forest floor. This, coupled with the loss of the litter layer, led to increased diversity and cover in the herbaceous layer and a higher frequency of serotinous pine seedlings. These conditions also resulted in increased soil temperature and decreased decomposition. The reduced rates of decomposition may facilitate the reformation of the litter and organic layer. The future community composition in the Linville Gorge is hard to predict, but will most likely be determined by the frequency, severity and intensity of future fires.

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