FINAL REPORT

Impacts of Repeated Wildfire on Vegetation in the Southern Appalachian Mountains

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JOINT FIRE SCIENCE PROGRAM

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Abstract

The infrequent occurrence of large wildfires in the southern Appalachian Mountains over the last several decades has offered few opportunities to study the impacts of these types of disturbances. As a result, relatively little is known about how heterogeneity in topography, vegetation, and recent disturbance history interact to influence patterns of fire severity across the landscape. Since 2000, five separate wildfires burned a large portion of the area in, and surrounding the Linville Gorge Wilderness in western North Carolina, two burned the same area a second time. Burn severity and vegetative recovery were measured in 152 plots established in 1992 prior to any wildfires. Field measurements of severity were strongly related to GIS data, allowing an accurate map of severity conditions in all 5 burned landscapes. Severity followed similar patterns at all fires. The most severely-burned areas were on the higher elevations, probably ridges, on slopes of moderate steepness, and facing to the south, southwest, west, or northwest. The effects of 1 burn vs. 2 burns were more pronounced in dry-site than in moist-site communities. Shrub and tree cover decreased at all levels of re-burn severity. Herb richness and total richness were highest in the highest level of re-burn severity. Significant changes (increases) in species richness occurred only at scales $< 1\ m^2$, which is likely a reflection of large-scale community heterogeneity and/or heterogeneous patterns of burn severity. Woody plant community assembly was not substantially affected by burning, nor did burning appear to promote exotic plant invasion. Our measurements and maps of fire severity will help managers to better understand vegetative response to wildfire across widely diverse topographic and severity conditions.

Background and Purpose

The acreage of land burned annually in the Appalachian region has increased steadily over the last thirty years with the rate of increase nearly doubling in recent years (Lafon et al. 2005). Projected increases in fire activity in the southeast due to climate change strongly suggest the area of recently burned land in the southern Appalachian Mountains is likely to continue increasing (Bachelet et al. 2001). As a result there will be a growing need for managers to understand the effects of fires returning to recently burned stands in relation to how repeated fire affects management goals. Information on re-burns in the southern Appalachian Mountains is limited (Arthur et al. 1998) and managers in the region can only use inference from past studies of re-burns in the western US for insights into effects of previous fires on re-burn severity (Thompson et al. 2007, Romme 1982). With little information available from within the region, managers of forests in the southern Appalachian Mountains have a need to understand the effects of repeated fire on woody regeneration, understory diversity, and invasion of exotic species.

Fire historically increased landscape heterogeneity in the southern Appalachian Mountains by maintaining a variety of community types across the landscape (Delcourt and Delcourt 1998). Anthropogenic fires perpetuated oak and chestnut (*Castanea dentata*) on upper slopes and fire adapted species such as Table Mountain pine and pitch pine (*Pinus rigida*) on drier ridges. A long period of fire exclusion has left many forest communities in a degraded state due to encroachment of more fire sensitive species (Williams 1998). Active prescribed fire programs began in the region as late as the 1980's and have gradually increased in importance since (Waldrop et al. 2007).

Although some research has been done on the effects of fire in the southern Appalachian Mountains, the information available to managers has been largely derived from the results of single relatively small prescribed fires intended to restore decadent Table Mountain pine (*Pinus pungens*) stands, promote oak (*Quercus* spp.) regeneration, and increase understory diversity and productivity (Turrill-Welch and Waldrop 2001, Kuddes-Fisher and Arthur 2002, Elliott et al. 1999) The rare occurrence of large wildfires in the region (Lafon et al. 2005) has offered few opportunities to study the effects of this type of disturbance at the landscape scale. In the fall of 2000 a wildfire burned the majority of Linville Gorge in western North Carolina after a prolonged drought and infestation of southern pine beetle. The existence of an extensive network of vegetation plots established prior to the 2000 fire enabled researchers to study how wildfire maintains both community and landscape level diversity (Reilly et al. 2006a, Reilly et al. 2006b), as well as the influences of topography and forest type on spatial patterns of fire severity (Wimberly and Reilly 2007). In the spring of 2007 two separate wildfires burned a portion of the landscape that burned in 2000 as well as much of the remaining unburned area within and surrounding Linville Gorge. This rare event was followed by another wildfire in 2008 that burned previously unburned areas of the Gorge. The occurrence of these fires provided a unique opportunity to follow up on previous work by re-measuring an extensive network of plots to examine the impacts of multiple disturbances on a number of biological variables of interest to land managers.

Studies of prescribed fire have documented effects on vegetation structure, woody regeneration of pine and oak species, and diversity in the understory (Turrill-Welch and Waldrop 2001, Kuddes-Fisher and Arthur 2002, Elliott et al. 1999, Waldrop et al. 2007). Table Mountain pine, a fire dependent species endemic to the southern Appalachians, has been of particular interest (Waldrop and Brose 1999, Welch et al. 2000, Turrill-Welch and Waldrop 2001). Studies on this species have elucidated microsite conditions and levels of fire severity necessary for seedling establishment. Effects of repeated fire in regenerating Table Mountain pine stands are unknown. In oak dominated forests, fire is thought to promote regeneration by reducing the density of undesirable hardwood saplings and midstory shrubs and increasing light levels (Van Lear and Watt 1993). However, studies on the application of single prescribed fires in oak forests found little effect on oak regeneration and species composition in the understory due to vigorous sprouting of other hardwood species and shrubs and suggest that repeated burning or fires of higher intensity may be necessary (Kuddes-Fisher and Arthur 2002).

A rare study of large wildfires in the southern Appalachian Mountains was done in Linville Gorge after a wildfire in 2000. Fire increased herbaceous species richness at intermediate scales ($\geq 100m^2$), but increases in tree species richness were limited to small scales ($<10m^2$) due to lack of immigration (Reilly et al. 2006a). Severity was correlated with the same environmental gradients structuring spatial patterns of community composition across the landscape and ultimately maintained species turnover and β-diversity (Reilly et al. 2006b). Spatial analysis of remotely sensed images found that patterns of fire severity were strongly constrained by forest type and topography (Wimberly and Reilly 2007). These results support the notion that fire promotes landscape heterogeneity, but also suggest that woody regeneration in areas of high tree mortality could be inhibited by strong dispersal limitation in dominant tree species.

Current knowledge of fire effects suggests two possible outcomes dependent on fire severity in re-burned areas. Information on re-burns is currently limited (Arthur et al. 1998) but past studies on areas subject to re-burn in the western US provide some insight on potential burn severity and effects in Linville Gorge. Despite contrasting results, these studies agree that fire severity depends on the potential of young regeneration to carry fire (Thompson et al. 2007, Romme 1982). The abundant pine regeneration and vigorous sprouting of hardwood trees and ericaceous shrubs typical after fire in the southern Appalachians suggest that fuels are likely capable of carrying severe fire. Re-burn severity in recently regenerated conifer stands in Oregon was higher than that of the initial burn (Thompson et al. 2007). If this is the outcome of the 2007 and 2008 fires and young Table Mountain pine regeneration is destroyed without having produced sufficient cones, subsequent regeneration will depend on input of seeds from outside areas. Likewise, if oak trees are killed by higher severity fire in the re-burned areas, regeneration will also depend on immigration of acorns. These outcomes could ultimately affect landscape diversity and productivity. However, if Table Mountain pines have sufficient seed stock and mature oaks survive; higher severity re-burns may be more successful than initial burns at eliminating understory shrubs and saplings and reducing litter and duff depth. These effects would further promote regeneration of Table Mountain pine and desired hardwoods as well as promoting diversity in the understory. Mountain laurel grows in dense thickets and represents a major source of vertical fuels, occasionally supporting crown fires (Waldrop and Brose 1999). Control of this species would reduce wildfire hazard and promote regeneration of desirable species.

This project was conducted to determine how fire severity and fire effects differ among areas with different recent burn histories. Specific objectives included:

1. To compare fire effects and spatial patterns of severity between areas measured before burning and after one (2000 and 2008) or two (2007 or 2008) wildfires,
2. To assess the effects of re-burn severity on woody regeneration (Table Mountain pine and oak species), understory species diversity, and exotic species invasion, and
3. To examine successional change in areas with different recent disturbance histories and effects on species turnover and diversity across the landscape.

Study Description and Location

Study Site

Linville Gorge is in the Pisgah National Forest south of Boone, NC and contains a 4390 ha federally designated wilderness area. Elevations range from 320 m at the bottom of the gorge to 1250 m on the upper slopes of the ridges. Topography is complex and the presence of steep environmental gradients has resulted in an extremely diverse landscape, much of which has never been logged and is thought to be representative of pre-settlement forests (Newell and Peet 1998). Upper slopes and ridges are dominated by thermic pine and oak communities with a thick layer of ericaceous shrubs. Thermic pine forests are composed predominantly of Table Mountain pine, pitch pine, and Virginia pine (*Pinus virginiana*). Thermic oak forests are dominated by chestnut oak (*Quercus prinus*), scarlet oak (*Quercus coccinea*), and white oak (*Quercus alba*). Mountain laurel, a major component of live fuel, is abundant in both. Lower slopes and coves are dominated by chestnut oak, white pine (*Pinus strobus*) and eastern hemlock (*Tsuga canadensis*) with a thick layer of rhododendron (*Rhododendron* spp.) in the understory. Prior to 2000, the last widespread surface fires occurred in the early 1950's. In 2000 a wildfire burned approximately 4000 ha in and surrounding Linville Gorge (Table 1). Fire severity was heterogeneous across the landscape with high severity crown fires occurring on steep slopes and upper ridges and low severity surface fires occurring on mid-slopes and coves (Fig. 1a) (Wimberly and Reilly 2007). In spring 2007, two separate fires burned a large portion of the landscape previously burned in 2000 as well as much of the remaining unburned area surrounding Linville Gorge (Fig 1b). Another large wildfire occurred in 2008 and burned much of the area immediately adjacent the area that burned in 2000 and 2007.

Table 1. Area impacted by 0, 1, or 2 wildfires.

Fire History	Area (ha)
Unburned	2,031
Burned once in 2000	1,632
Burned once in 2007 or 2008	1,602
Burned twice, 2000 and 2007	2,477

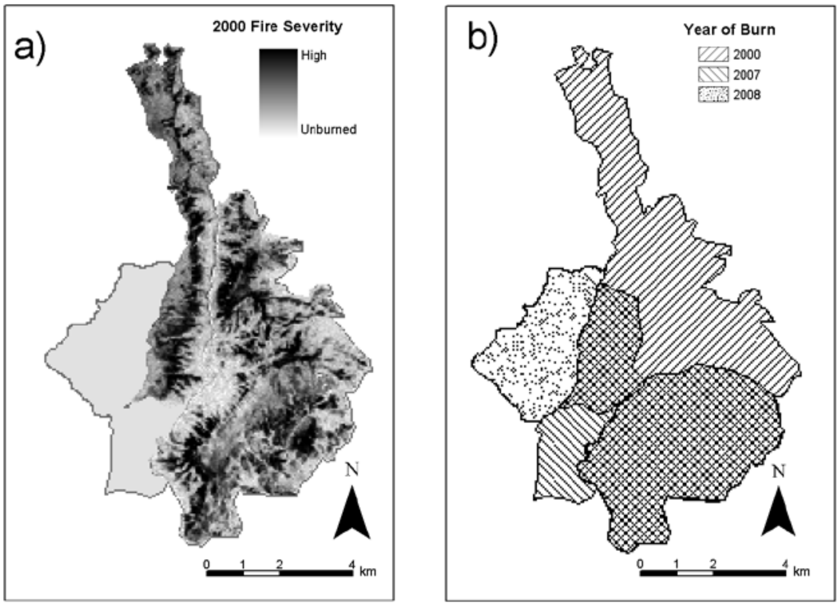


Figure 1. Linville Gorge maps showing a) fire severity in 2000 (from Wimberly and Reilly 2007) and b) approximate boundaries of the 2000, 2007, and 2008 wildfires.

Sampling

Plots were characterized as burned twice, burned once in 2000, burned once in 2007 or 2008, and unburned. We re-measured two previously established sets of plots and established new plots to increase sample size and spatial coverage in underrepresented areas. The effects of fire on plant community assembly were assessed using Analysis of Variance (ANOVA) and Nonmetric Multidimensional Scaling (NMS).

The first set of previously-measured plots consists of a series of 181 intensively sampled preburn vegetation plots established in 1992 to document the variation in composition and structure of vegetation in Linville Gorge (Newell and Peet 1998). Plots were stratified to capture topographic and geologic variation and were sampled following the North Carolina Vegetation Survey (NCVS) protocols (Peet et al. 1998). This sampling methodology involved intensive sampling of the understory in a series of nested subplots ranging from 0.01 m^2 to 10 m^2 within one to four adjacent 100 m^2 modules. All species were counted and assigned cover estimates at the 100 m^2 scale. Woody vegetation $\geq$1.4 m diameter at breast height was sampled within each 100 m^2 module.

The second set of plots was a series of 57 rapid assessment plots established in 2003 to capture the spatial variation of fire severity across the landscape. A modified version of Key and Benson (2002) was used to create a Composite Burn Index (CBI). This was based on a visual estimate of the percent mortality of dominant canopy trees and mid-canopy trees within a 30 m radius and correlated strongly with changes in basal area (Wimberly and Reilly 2007). Of these plots, 49 burned once in 2000 and 8 burned twice, 2000 and 2007. All rapid assessment plots were measured after the 2000 wildfire.

We re-measured the majority of NCVS plots using the same measurements that were taken in 1992. Plot re-measurement was determined by the number of plot markers that were found and the patterns of fires that burned through the plots. Our goal was to measure 50 or more plots that had been burned once, twice, or not at all (Table 2). Special emphasis was given to find and measure those plots that were measured both before and after the 2000 wildfire. Field crews spent approximately 8,500 person hours locating and measuring the NCVS plots over the growing seasons of 2009, 2010, and 2011. Decay of reference trees, inaccurate GPS points from old equipment, and loss of monuments since 1992 contributed to the loss of many of the original plots. However, we exceeded our goal of finding and measuring 150 plots with a total of 154. Of those, 53 were unburned, 51 were burned once, and 50 were burned twice.

Table 2. Number of plots previously measured and re-measured by fire history and sample type.

Plot Type and Burn History	Measured in 1992 (pre-burn)	Measured in 2003 (after first wildfire)	Re-measurement
NCVS Intensive Sample Plots			
Unburned	58	11	53
Burned once in 2000	45	14	14
Burned once in 2007 or 2008	23	0	37
Burned twice, 2000 and 2007	55	21	50
Rapid Assessment Plots			
Unburned	0	0	0
Burned once in 2000	0	49	49
Burned once in 2008	0	0	26
Burned twice, 2000 and 2007	0	8	8

Additional response variables were measured on NCVS plots at the module scale (100 m^2) and in three 1 m^2 subplots following protocols of Waldrop and Brose (1999). Severity in subplots was estimated using the following categories: 1) unburned, 2) burned with partially consumed litter present, 3) no litter present and 100% of the area covered by duff, 4) soil exposure on 1-30% of the area, 5) soil exposure on 31-60% of the area, and 6) soil exposure on 61-100% of the area. Insolation was estimated between 10:00 hours and 14:00 hours on sunny days and described as one of the following categories: 1) Full shade, 2) 1-30% of the area receiving direct sunlight, 3) 31-60% of the area receiving direct sunlight, and 4) 61-100% of the area receiving direct sunlight. Density of all seedlings and hardwood sprouts were counted in each 1 m^2 subplot by species. Litter and duff depths were taken at one corner of each subplot. At the module level, severity and insolation was estimated again, as well as cover of litter, duff, mineral soil, rock, and downed wood.

All 57 of the rapid assessment plots were re-measured using the same procedures as before. An additional 26 plots were established in the area burned once in 2008. Multi-temporal Landsat Imagery was used to investigate spatial patterns of fire severity across the landscape. Fire severity was estimated using the difference in the Normalized Burn Index (*d*NBR) between pre- and post-fire imagery. Spatial data for *d*NBR from the 2000 fire already exist, so only the pre- and post-fire imagery had to be acquired from the USGS for the 2007 and 2008 wildfires.

Data Analysis

Objective 1- Comparing initial fire effects and spatial patterns of severity between areas subject to initial burn and re-burn in the 2007 or 2008 fires.

Initial fire effects were compared using an ANOVA design. Comparing fire severity in re-burned areas to those burned once followed Thompson et al. (2007). This method used the dNBR from pre- and post-fire Landsat imagery and a series of random points which were incorporated into generalized least squares regression models. Assessing the roles of topography and forest type between the two areas followed Wimberly and Reilly (2007). Accuracy of the dNBR values was estimated using CBI values calculated from field plots using a simultaneous autoregressive nonlinear regression model that explicitly incorporates spatial autocorrelation. This model was used to create a spatially explicit map of predicted CBI values to be used with a digital elevation model and detailed vegetation map (Newell and Peet 1998) to compare the roles of topographic variables and forest type in determining fire severity between the initial fire and re-burn.

We obtained pre-and post-fire Landsat imagery to calculate the Relativized difference in the Normalized Burn Ratio (R*d*NBR) (Miller and Thode 2007) for each of the wildfires in 2007 and 2008. The best available images used to calculate the R*d*NBR from the 2007 fires were from 6/10/2006 and 8/16/2007. The best available images used to calculate R*d*NBR for the 2008 fire were from 8/16/2007 and 9/3/2008. We created a standardized composite burn index (CBI) to calibrate the R*d*NBR values from the 2007 fires and the 2008 fire as opposed to comparing them directly due the lack of availability of imagery on anniversary dates and the resulting differences in time since fire. We attempted to remove the bias due to different time since fire by using different models for each year to predict an index that has been calibrated to actual fire effects.

We used a modified version of the Fire Effects Inventory and Monitoring (FIREMON) protocol (Key and Benson 2002) to visually estimate fire effects in four categories within a 15 meter radius of plot center on a 300 point scale. The four categories included % duff consumption, % midstory and shrub topkill, % mortality of midstory trees, and % mortality of overstory trees. These particular metrics were chosen because they were all measured consistently across the various sources of field data, and because they could be reliably measured 3–4 years after the fire. These metrics were recorded as index values ranging between 0 and 3, with higher numbers indicating greater mortality (Key & Benson, 2002), and the CBI was a computed as a weighted average with weights of 1, 2, and 3 assigned to the shrub/small tree, intermediate tree, and large tree strata. This weighting put more emphasis on the mortality of larger trees, which were the most reliable field-based indicators of long-term fire effects. The composite CBI values were rescaled to range between 0 and 1, with unburned plots assigned a CBI value of zero. Each plot was associated with the *d*NBR measurement from the nearest pixel by overlaying plot centroids on the *d*NBR image. Non-linear regression was used to fit an equation relating the observed CBI indices to *d*NBR, and this equation was then used to predict CBI for all pixels falling inside the fire perimeter. All pixels in each wildfire were then sampled for CBI, elevation, topographic moisture, slope, and aspect. We compared the distributions of CBI values across each of these gradients.

Objective 2- Assessing the effects of re-burn severity on woody regeneration, understory species diversity (richness and species turnover), and exotic species invasion.

Data analysis for Objective 2 was similar to Reilly et al. (2006a) but required a repeated measures approach instead of a paired approach. Pre-fire data at the species level was limited to cover at the module scale, but density of woody regeneration was estimated from subplots to allow an assessment of how well cover predicts density. The NCVS sampling methodology allows examination of species richness at multiple spatial scales within a plot (Peet et al. 1998). Species-area curves were created using non-linear regression and coefficients (small scale richness) and slopes (turnover within the plot) were compared for changes. These were correlated with fire severity as estimated from changes in basal area to assess the role of these processes and compare their relative importance after re-burn to that of initial burn. The six severity classes measured in the field were combined to three severity classes: low (unburned to burned with partially consumed litter present), medium (no litter present and 100% of the area covered by duff to soil exposure on 1-30% of the area) and high (soil exposure on 31-100% of the area). Plots were described by community types and repeated measures ANOVA was used to test for changes among the three sampling periods in all response variables. Since only a fraction of study plots burned twice, data for thermic oak and acidic cove communities were pooled.

Objective 3- Examining successional change in areas with different recent disturbance histories and compare species turnover and diversity across the landscape from 1992 to 2009.

Analyses for this objective followed those used in Reilly et al. (2006b). Nonmetric Multidimensional Scaling was used to ordinate plots and to determine compositional similarity.

Key Findings

Finding 1

Both R*d*NBR data from remote sensing and a composite burn index (CBI) that we derived from field data proved to be exceptionally accurate at predicting on-the-ground differences in fire severity. This allowed us to use GIS data to produce an accurate map of fire severity in three landscapes that were burned once and two landscapes that were burned twice.

All wildfires resulted in extremely heterogeneous patterns of fire severity across the landscape ranging from low levels of litter consumption to total litter and duff consumption and complete midstory and canopy mortality. R*d*NBR values were strongly correlated with field based estimates of fire effects as well as CBI values (Table 3). When possible, we compared results among 5 different burn units: Dobson Knob (burned once), Shortoff Mountain (burned twice), Pinnacle (burned once), Pinnacle (burned twice), and Sunrise (burned once) (Fig. 2).

Table 3. Relationship of R*d*NBR values to field measure of fire severity and the Composite Burn Index.

Variable	R^2 for plots burned in 2007	R^2 for plots burned in 2008
% Duff Consumption	0.55	0.60
% Midstory and Shrub Topkill	0.62	0.69
% Mortality of Subcanopy Trees	0.50	0.55
% Mortality of Canopy Trees	0.70	0.73
CBI	0.76	0.78
CBI Model	y=1.58x+224	y=1.33x+156

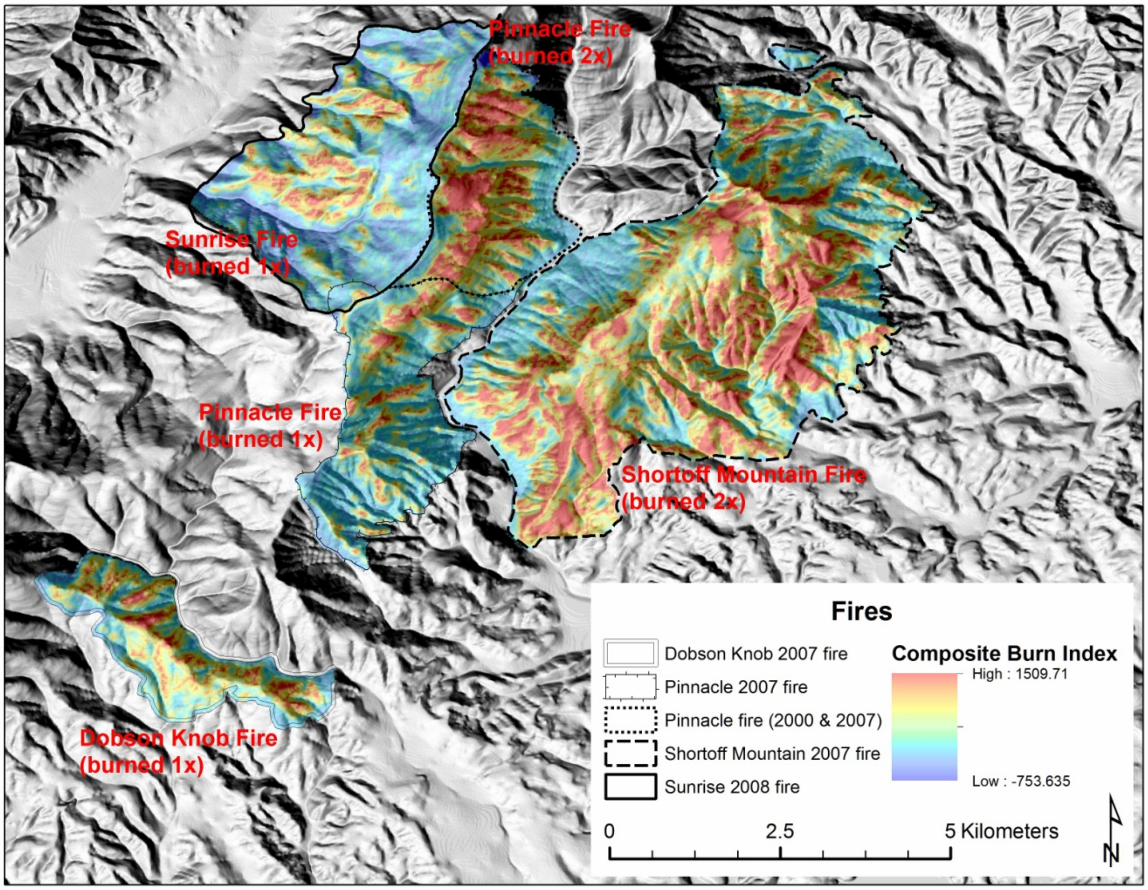


Fig. 2. Predicted Composite Burn Index (CBI) values for the 2007 and 2008 wildfires in and around Linville Gorge.

<u>Finding 2</u>

CBI followed similar patterns at all fires. The most severely-burned areas were on the higher elevations, probably ridges, on slopes of moderate steepness, and facing to the south, southwest, west, or northwest.

The distributions of CBI values over major environmental gradients were similar among each of the five landscapes (Fig. 2), although they were highly variable within each landscape. CBI values were generally higher in landscapes subject to repeated fire, particularly the Shortoff Mountain Fire. CBI values within the area burned twice in the Pinnacle Fire were higher than those in burned once in the Pinnacle Fire and Sunrise Fire. CBI values within the area burned once in the Dobson Knob Fire area were similar to those burned twice in the Pinnacle Fire except over the elevational gradient where those burned twice in the Pinnacle Fire were higher.

CBI values tended to be highest at the highest elevations where ridges burned severely (3,000 to 3,500 feet) but there were no statistical differences (Fig 3a). Fires were less severe in areas with higher TMI values but there were no significant differences. TMI values were over 800 if TMI was less than 1, but were approximately 700 if TMI was greater than 1 (Fig. 3b). Severity was greatest on moderate slopes as compared to flatter or steeper slopes (Fig 3c). CBI was approximately 800 at slopes of 30 to 60 percent but only about 650 on slopes at 15-30 percent and 60-75 percent. Aspect was most closely related to post-burn severity (Fig. 3d). Although there was high variability, CBI was significantly lower on north-facing slopes (425) than on west- and southwest-facing slopes (975). Other slopes that had conditions of high severity included those facing to the south and northwest.

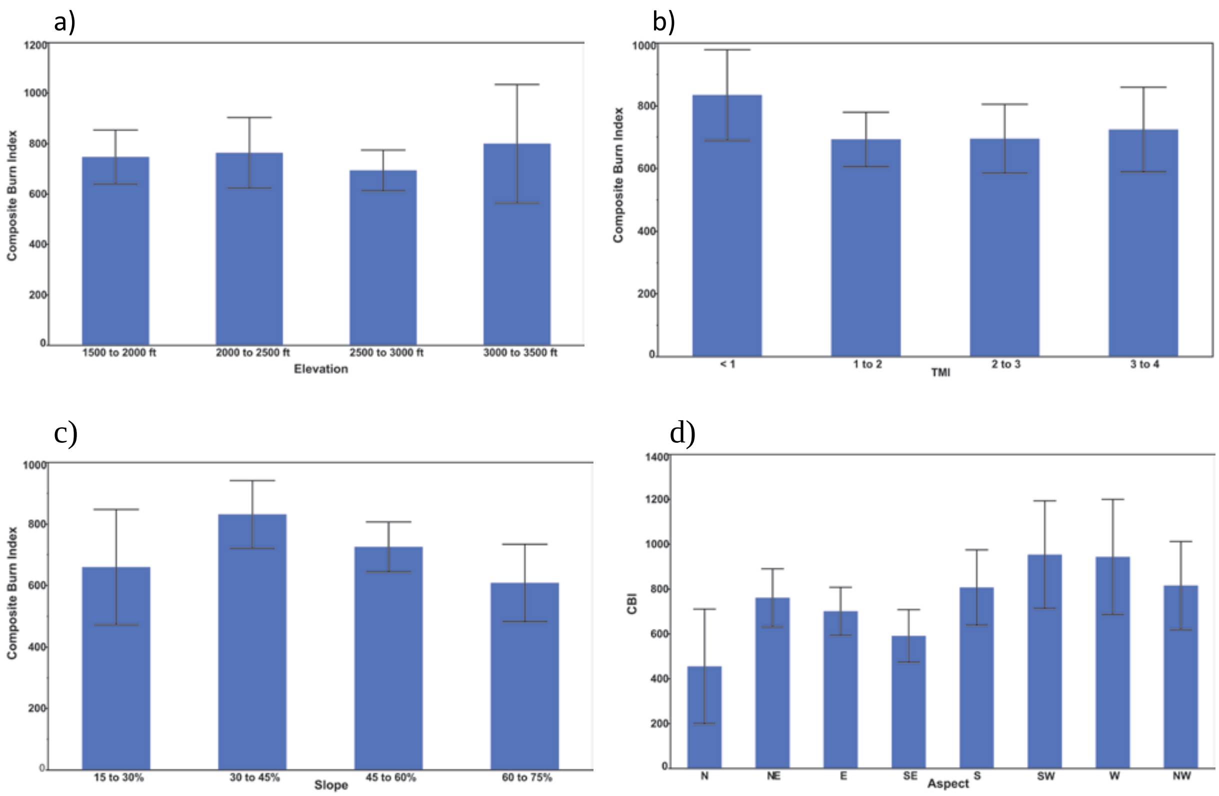


Fig. 3. The distribution of CBI values across a) elevation, b) topographic moisture, c) slope, and d) aspect for the 2007 and 2008 wildfires.

The distribution of CBI values over environmental gradients was similar in areas burned both once and twice. Conditions of high severity were most common on ridges and south and west facing slopes. These results suggest that landscape heterogeneity is a major driver of spatial patterns of fire severity regardless of fire history and cover type. Elevation was not strongly related to severity, although the highest elevations had the most severe fires. This trend may have occurred as fires ran upslope, became intense on ridges, and residence time was long because fires could not move upslope. Low-severity fires tended to occur on slopes that were either flat or extremely sharp. Flat slopes may have also been those that were of lowest elevation and moist, producing cooler and less severe fires. Steep slopes, on the other hand, supported fast-moving fires that moved through an area without consuming large portions of the forest floor. As should be expected, moist sites burned at lower severity than did dry sites. Although not significant, there seemed to be a threshold value at TMI=1 where severity, as expressed by CBI, was higher at lower TMI values but unchanged as moisture increased.

Finding 3

The effects of 1 burn vs. 2 burns were more pronounced in the thermic oak community than in the acidic cove community. Shrub and tree cover decreased at all levels of re-burn severity. Herb richness and total richness were highest in the highest level of re-burn severity.

The strong relationship of fire severity to topography provided a problem for analysis because topography had to be included. However, the highly diverse topography of Linville Gorge provided an almost infinite number of site and cover types. Therefore, we chose to focus our analyses of plant response to two site type at either extreme of the moisture gradient, moist acidic coves and dry thermic oak sites. Acidic cove communities occupied lower slopes below bluffs and were dominated by chestnut oak (*Quercus prinus*), red maple (*Acer rubrum*), eastern white pine (*Pinus strobus*), Carolina hemlock (*Tsuga canadensis*), great rhododendron (*Rhododendron maximum*), and associated species. Thermic oak forests occupied upper slopes or ridges and were dominated by chestnut oak, white oak (*Quercus alba*), scarlet oak (*Quercus coccinea*), Eastern hemlock (*T. Canadensis*) and various pine species.

At the plot level, the effects of fire on plant cover varied by community type (acidic cove or thermic oak), and were – in some cases – influenced by the number of burns. For acidic cove communities, decreases in total plant cover were only observed in unburned plots (Table 4). While total plant cover was not influenced by fire in acidic cove plots, the relative contributions of some growth forms did change after burning. Graminoid and herb cover, for example, increased approximately five-fold in plots burned twice. Herb cover increased more than four-fold in plots burned once. No such effects were observed for shrubs, trees and vines in acidic cove plots. In the thermic oak community, significant decreases in total cover occurred in all burn categories. In plots burned twice, there was an eleven-fold increase in graminoid cover and large decreases in tree and shrub cover. Decreases in tree cover also occurred in plots burned once.

Burning also influenced species richness (Table 4). In acidic cove plots, nearly two-fold increases in total species richness were observed in plots burned once and twice, largely due to substantial increases in shrub, herb and tree richness. Similar patterns were observed in the thermic oak community, although the effect of one burn versus two burns was not evident. Species richness increased by over 40% in plots burned once and nearly 74% in plots burned twice. These increases corresponded with increases in the richness of graminoids, herbs, trees and vines. Shrubs were the only growth form for which burning appeared to have no influence on species richness.

Largely in concert with the above effects on cover and richness, burning also influenced diversity indices (Table 4). For the acidic cove community type, the effects were more pronounced for the Shannon index than they were for the Simpson's index. Significant increases in Shannon diversity for all species combined were observed in plots burned both once and twice. Graminoid diversity (Shannon and Simpson) increased in plots burned twice and Shannon diversity for shrubs increased in plots burned once. For the thermic oak community type, significant increases in Shannon and Simpson diversity occurred for all species combined, graminoids, herbs and shrubs. Significant increases in Shannon and Simpson diversity for vines were observed in plots burned once.

Table 4. Before and after comparisons of cover, species richness, Shannon Diversity and Simpson's diversity (by community type and growth form) in unburned sites and sites burned 1x or 2x. Before and after pairs in bold and highlighted are statistically different at alpha = 0.05.

			Cover		Richness		Shannon		Simpson	
			Before	After	Before	After	Before	After	Before	After
Acidic Cove	All species	Unburned	**3.415**	**2.371**	16.538	17.077	1.679	1.692	0.730	0.713
		Burned 1x	2.443	2.029	**18.455**	**31.182**	**1.795**	**2.405**	0.755	0.830
		Burned 2x	2.172	1.541	**24.57**	**48.001**	**2.092**	**2.820**	0.816	0.847
	Graminoids	Unburned	0.001	0.002	0.154	0.154	0.000	0.000	0.000	0.000
		Burned 1x	0.008	0.024	1.091	1.727	0.232	0.420	0.135	0.242
		Burned 2x	**0.024**	**0.114**	**1.142**	**5.000**	**0.128**	**1.161**	**0.061**	**0.590**
	Herbs	Unburned	0.009	0.035	5.615	5.846	1.324	1.361	0.611	0.630
		Burned 1x	**0.020**	**0.082**	7.091	12.364	1.286	1.999	0.570	0.769
		Burned 2x	**0.017**	**0.119**	**11.429**	**22.429**	1.860	2.507	0.764	0.860
	Shrubs	Unburned	1.261	1.034	3.692	4.308	0.463	0.443	0.254	0.223
		Burned 1x	0.671	0.526	**3.455**	**7.091**	**0.594**	**1.196**	0.322	0.559
		Burned 2x	0.597	0.425	**3.857**	**7.286**	0.655	1.261	0.356	0.556
	Trees	Unburned	2.123	1.300	6.769	6.769	1.477	1.301	0.725	0.658
		Burned 1x	1.744	1.397	**6.818**	**10.000**	1.382	1.627	0.679	0.714
		Burned 2x	1.534	0.883	**8.142**	**13.286**	1.582	1.879	0.745	0.786
	Vines	Unburned	0.021	0.000	0.308	0.000	0.067	0.000	0.037	0.000
		Burned 1x	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
		Burned 2x	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Thermic Oak	All species	Unburned	**3.721**	**2.781**	34.154	30.231	0.679	0.996	0.302	0.465
		Burned 1x	**4.017**	**2.175**	**26**	**38.571**	0.681	1.294	0.328	0.500
		Burned 2x	**2.658**	**1.332**	**29.83**	**49.347**	**0.981**	**2.569**	**0.447**	**0.834**
	Graminoids	Unburned	0.001	0.001	0.308	0.769	0.055	0.035	0.033	0.025
		Burned 1x	0.000	0.012	**0.143**	**1.571**	0.000	0.406	0.000	0.232
		Burned 2x	**0.010**	**0.110**	**0.450**	**1.965**	**0.303**	**1.298**	**0.181**	**0.603**
	Herbs	Unburned	0.103	0.058	9.692	8.462	0.874	1.201	0.387	0.541
		Burned 1x	0.107	0.110	**5.500**	**10.643**	0.752	1.364	0.356	0.537
		Burned 2x	0.071	0.095	**6.571**	**20.286**	**1.010**	**2.481**	**0.465**	**0.836**
	Shrubs	Unburned	1.121	0.960	9.923	7.077	1.138	0.969	0.532	0.463
		Burned 1x	1.194	0.725	8.429	9.929	1.230	1.456	0.577	0.610
		Burned 2x	**0.856**	**0.257**	7.952	8.810	**1.264**	**1.665**	**0.584**	**0.718**
	Trees	Unburned	2.303	1.646	12.462	12.000	1.847	1.776	0.783	0.767
		Burned 1x	**2.602**	**1.141**	**10.357**	**14.214**	1.793	1.755	0.774	0.730
		Burned 2x	**1.560**	**0.673**	**12.714**	**15.619**	1.957	1.920	0.812	0.773
	Vines	Unburned	0.193	0.116	1.769	1.923	0.463	0.547	0.322	0.384
		Burned 1x	0.114	0.187	**1.571**	**2.214**	**0.298**	**0.668**	**0.197**	**0.434**
		Burned 2x	0.161	0.197	**2.143**	**2.667**	0.520	0.735	0.350	0.03

In study areas burned a second time, patterns of severity along these gradients were almost always higher than in areas burned once (Pinnacle 1x vs. Pinnacle 2x / Dobson Knob 1x vs.

Shortoff Mountain 2x). CBI values from the Sunrise Fire were similar to those burned once in the Pinnacle Fire but generally a little lower than those burned in the Dobson Knob Fire. We propose that severity is higher in areas burned a second time because areas burned recently have greater insolation and higher temperatures, increased amounts of dead and downed woody fuels, abundant regeneration of conifers, and a dense layer of sprouting ericaceous vegetation. If the second fire is soon enough after the first, the forest floor may not have had sufficient time to reach preburn levels, thus exposing soil and creating conditions of greater severity.

Likely due to their differences in landscape position and pre-fire vegetative composition, as described above, thermic oak communities generally exhibited a greater response to burning than did acidic cove communities. These responses were most pronounced in areas subjected to two burns, as we observed significant changes in total cover, graminoid cover, shrub cover, and tree cover in these plots. They corresponded to alterations (increases) in plant species richness and diversity in these same plots. The growth form types affected most positively by repeated fire were graminoids and herbs, which is not surprising considering that these groups were largely comprised of ruderals and ephemerals which would be expected to respond quickly – if only temporarily – to such a disturbance. Fire severity correlates strongly with decreases in overstory basal area (Wimberly and Reilly 2007) due to the mortality of overstory trees. These decreases in tree cover, coupled with reductions in shrub cover, likely facilitated the establishment of these light-demanding species (Arthur et al. 1998; Kuddes-Fisher and Arthur 2002). Thermic oak plots burned once responded in a similar fashion to those burned twice, but the changes were generally of smaller magnitude (and often statistically insignificant). These findings suggest that single or once-repeated burns, regardless of severity, may not significantly alter thermic oak plant assemblages in the long term. These findings corroborate observations from other studies (Arthur et al. 1998; Elliott et al. 1999).

Response to fire was more variable in the acidic cove community, and the relationship with fire history (1x vs 2x) was less evident. This could be due to the fact that plots in this community generally burned at lower severity than those in the thermic oak community and/or reductions in statistical power due to a smaller sample size. In general, however, the largest changes in cover, richness and diversity for burned plots in this community type occurred for ruderal graminoids and herbs – the same pattern that was observed for the thermic oak community. This again suggests that such alterations, if they did occur, were temporary.

The significant reductions in tree cover in unburned plots, in both the acidic cove and thermic oak community, reflect substantial overstory mortality that occurred due a combination of combination of drought and pine beetle infestation during the study period.

Finding 4

Significant changes (increases) in species richness occurred only at scales < 1 m^2, which is likely a reflection of large-scale community heterogeneity and/or heterogeneous patterns of burn severity.

Species richness increased with increasing burn severity, but the significance of these effects was dependent on scale (Table 5). At the larger spatial scales (100 and 10 m^2) increases in richness – while substantial – were not statistically significant. At smaller spatial scales (1, 0.1 and 0.01 m^2), however, species richness in plots re-burned at high severity was significantly higher (1.21) than in those re-burned at low severity (0.04).

Table 5: Change in species richness at various spatial scales for plots re-burned at different severities in Linville Gorge Wilderness Area, North Carolina. Values for a given scale with different lower case letters are statistically different at alpha = 0.05.

	Change in species richness					
Scale (m^2)	**Low severity**		**Medium severity**		**High severity**	
100	15.11	a	23.67	a	25.61	a
10	7.58	a	11.12	a	14.61	a
1	2.56	b	4.62	ab	6.77	a
0.1	0.22	b	1.63	ab	2.21	a
0.01	0.04	b	0.71	ab	1.21	a

Comparisons of species area curves shed additional light on the relationship between re-burn severity and spatial scale. The slopes and intercepts of log-transformed curves were similar prior to the initial burn (Fig. 4). After the second burn, differentiation among the lines becomes readily apparent. All re-burned plots had similar slopes, but they had different intercepts. Plots burned by high-severity fires had the highest intercept, followed by those burned at medium and low severities. Divergence in richness from unburned plots increased with increasing spatial scale.

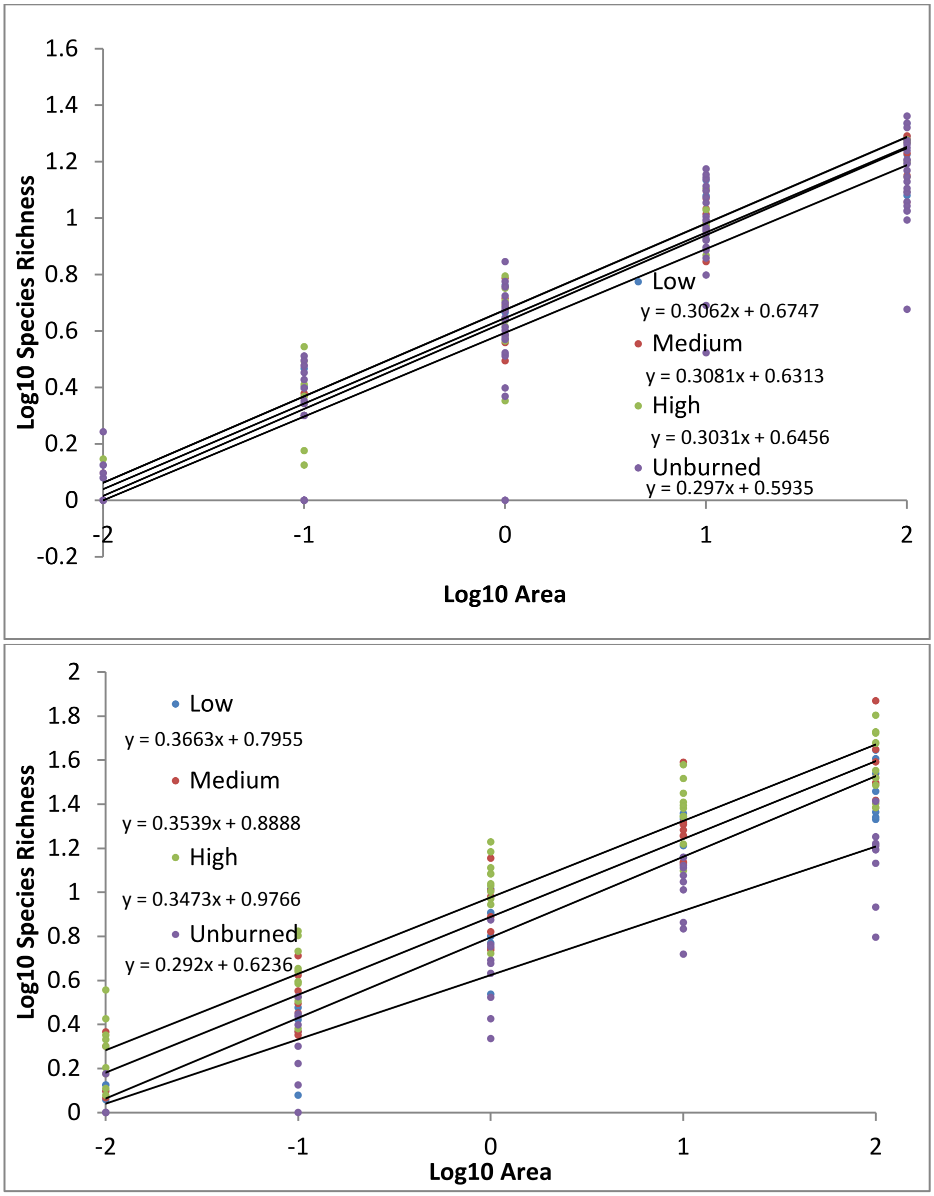
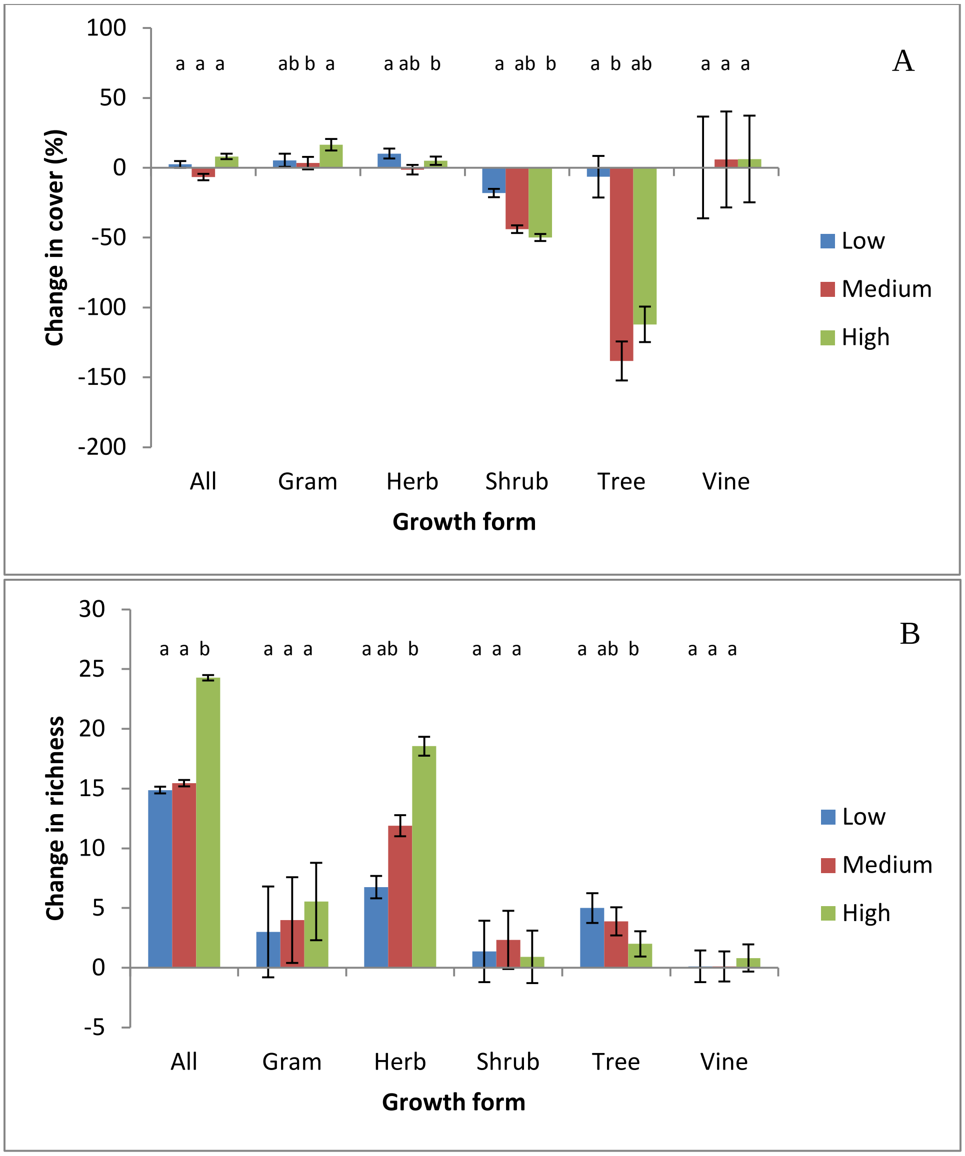


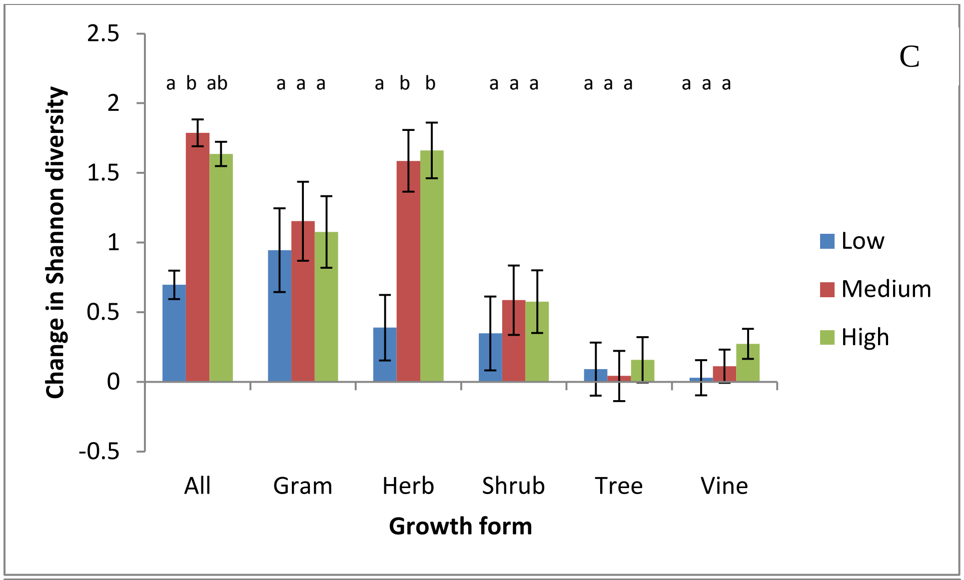
Figure 4. Log-transformed species area curves for plots re-burned at different degrees of severity in Linville Gorge Wilderness Area, North Carolina (Pre-fire: top, Post-fires bottom).

Total cover plant for all 3 burn severity categories did not change substantially following the second fire. However, substantial changes did occur for certain growth form types. Plots re-burned at medium or high severity, for example, saw more than double the decrease in shrub cover than those re-burned at low severity. The decrease in tree cover was even more dramatic. Graminoid cover increased the most in plots subjected to a high severity second burn, while herb cover increased most in the low severity re-burn plots. Vine cover increased significantly with increasing re-burn severity (Fig 5A). Following the second fire, total cover and tree cover were highest in low-severity reburn plots, although differences with other burn severity categories were not statistically significant. Post-fire graminoid and vine cover were highest in the highest burn severity class, and herb cover was highest in the lowest burn severity class. These differences were statistically significant (Fig. 6A).

Species richness increased at all levels of re-burn severity. However, the most substantial (and statistically significant) increases in total species richness occurred in plots subjected to a high severity second burn. These plots experienced an increase of approximately 10 more species than those burned at medium or low severity. For herbs, the magnitude of change increased significantly with increasing re-burn severity (Fig. 5B). Post-fire total species richness was highest in the highest re-burn severity category. This difference was primarily due to the high post-fire cover of herbs in these plots (Fig. 6B).

Re-burn severity also influenced species diversity. Increases in Shannon diversity were observed for all species combined, as well as for each of the five growth form types. The largest increases in Shannon diversity occurred for herbs, and these increases were more pronounced in plots re-burned at either medium or high severity. After the second burn, Shannon diversity for all species combined was highest in plots burned at high severity and this pattern held true for all growth forms except for graminoids. Herbs, however, were the only growth form for which a significant difference between burn categories was observed. Similar patterns were observed for the Simpson diversity index, with medium and high severity fires generally resulting in larger increases in diversity than low intensity fires. Graminoids were the only exception. Substantial increases for this growth form type were observed in all burn severity categories (Fig. 5C and 5D). Post-fire, Simpson diversity for all species combined was highest in medium and high intensity re-burn plots, and this pattern held true for most growth forms (Figs. 6C and 6D).


A
Change in cover (%)
100
50
0
-50
-100
-150
-200
a a a
ab b a
a ab b
a ab b
a b ab
a a a
All
Gram
Herb
Shrub
Tree
Vine
Growth form
Low
Medium
High
B
Change in richness
30
25
20
15
10
5
0
-5
a a b
a a a
a ab b
a a a
a ab b
a a a
All
Gram
Herb
Shrub
Tree
Vine
Growth form
Low
Medium
High



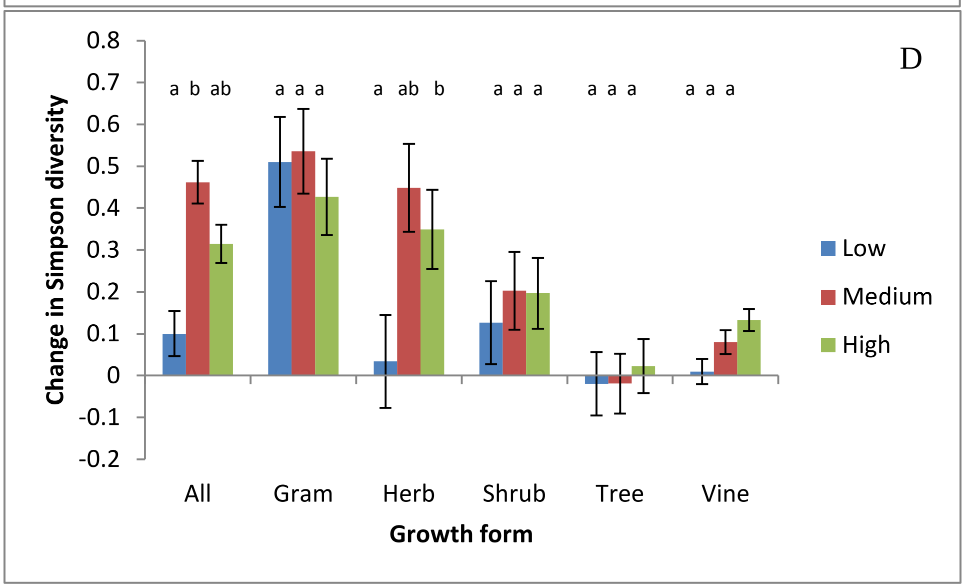
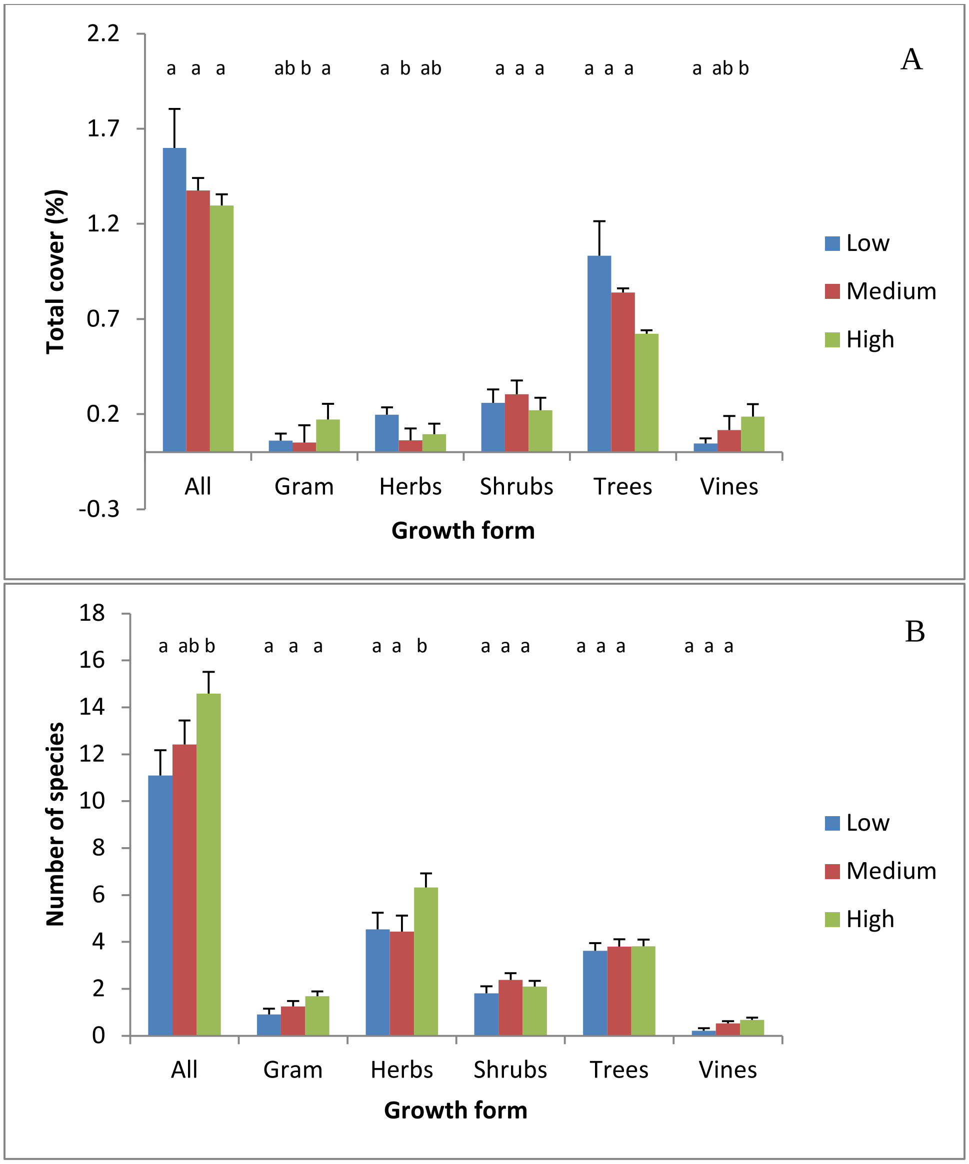


Figure 5: Percent changes in cover (A), richness (B), Shannon diversity (C) and Simpson's diversity (C) (by growth form) in plots re-burned at different degrees of severity (low, medium and high) in Linville Gorge Wilderness Area, North Carolina.


A
Total cover (%)
2.2
1.7
1.2
0.7
0.2
-0.3
a a a
ab b a
a b ab
a a a
a a a
a ab b
All
Gram
Herbs
Shrubs
Trees
Vines
Growth form
Low
Medium
High
B
Number of species
18
16
14
12
10
8
6
4
2
0
a ab b
a a a
a a b
a a a
a a a
a a a
All
Gram
Herbs
Shrubs
Trees
Vines
Growth form
Low
Medium
High

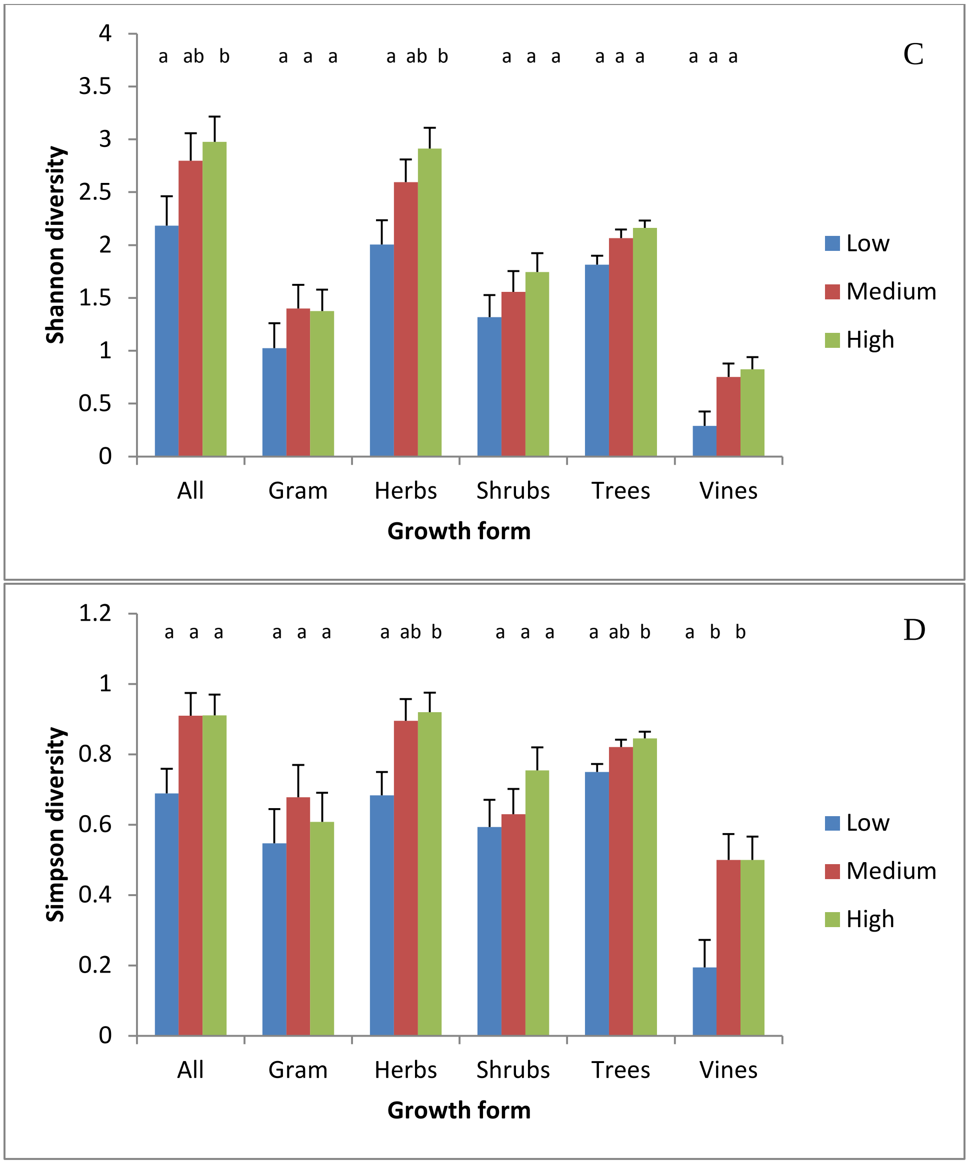


Figure 6. Current cover (A), species richness (B), Shannon Diversity (C) and Simpson diversity D) (by growth form type) in plots re-burned at different degrees of severity (low, medium and high) in Linville Gorge Wilderness Area, North Carolina.

The relationship between pre-fire shrub cover and post-fire tree cover in sites burned twice was weak. However, it appears that lower severity second fires tended to occur in sites with low pre-fire shrub cover (e.g. cover values <1). Post-fire, these plots generally had higher tree-cover than those with higher pre-fire shrub cover (e.g. cover values between 1 and 5), which tended to burn at medium or high severities (Fig. 7). There was no apparent relationship between re-burn severity on hardwood or Table Mountain pine regeneration, nor did re-burn severity appear to influence exotic plant invasion.

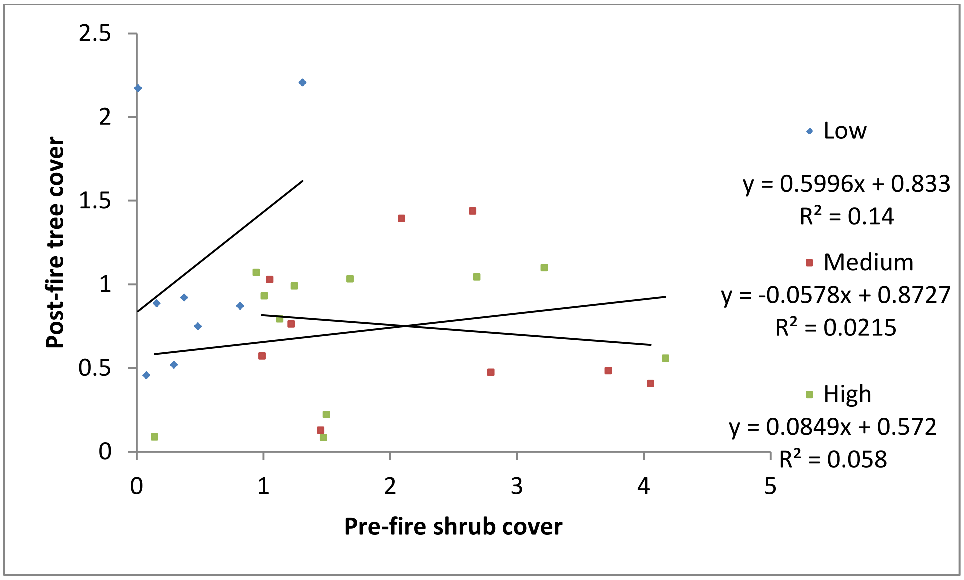


Figure 7: Regression analysis showing the relationship between pre-fire shrub cover and post fire (2x) tree cover in plots burned at different degrees of severity (low, medium and high), for shrubs (top) and trees (bottom) in Linville Gorge Wilderness Area, North Carolina.

At spatial scales of $1m^2$ or smaller, the increases in species richness for plots subjected to a high severity second burn are consistent with the observed increases in richness of small-statured non-woody plants in plots described above. In some plots, this trend also reflects an increase in tree and shrub species regeneration, as mineral soil exposure and increases in solar insolation caused by high severity fires created the requisite seedbed and light conditions for early- and mid-successional trees and shrubs to germinate from the seed bank. The inherent heterogeneity of these communities, which may have contributed to their heterogeneous patterns of re-burn severity, may explain why similar trends were not observed at larger scales. Species richness (including plots that never burned) did not differ substantially at any spatial scale prior to the first burn. The divergence between re-burned and unburned plots that occurred after the second burn reflects increases in large-scale community heterogeneity caused by burning (Reilly et al. 2006a; Reilly et al. 2006b).

Re-burning facilitated the establishment of graminoids and herbs – a pattern that was particularly evident in plots re-burned at medium or high severity. This again suggests that ruderals were the species group most adept at capitalizing on the pulse of resource availability caused by the fires (Arthur et al. 1998). Increases in tree and shrub richness and diversity, despite decreases in cover for these two growth forms, suggest that burning promoted woody regeneration as well. A relationship between woody regeneration and the different levels re-burn severity, however, was not clearly evident.

The relationship between pre-fire shrub cover and post-fire tree shrub cover suggests that shrubs influence fire severity. Shrubs, especially species such as mountain laurel (*Kalmia latifolia*), are a major component of the live fuels in these community types. They often act as ladder fuels, which can increase flame heights and kill or cause damage to overstory trees (Stottlemeyer et al.

2006). This likely explains why plots with low pre-fire shrub cover burned at lower severities and had higher post-fire tree cover than those burned at medium or high severity.

Finding 5

Woody plant community assembly was not substantially affected by burning, nor did burning appear to promote exotic plant invasion.

Nonmetric Multidimensional Scaling outputs of plots burned once or twice, relative to unburned plots (Fig. 8) do not strongly indicate that fires had substantial effects on woody plant composition. Despite the significant differences in cover, richness and diversity reported above, point clouds for each of the 3 burn classes largely overlapped. Some separation, however, occurred along ordination axes due to differences in the relative covers a handful of common species. For axis 1, the most influential species were *Acer rubrum* and *Nyssa sylvatica* . For axis 2 they were *Kalmia latifolia* and *Vaccinium pallidum*, and for axis 3 they were *Acer rubrum* and *Kalmia latifolia*.

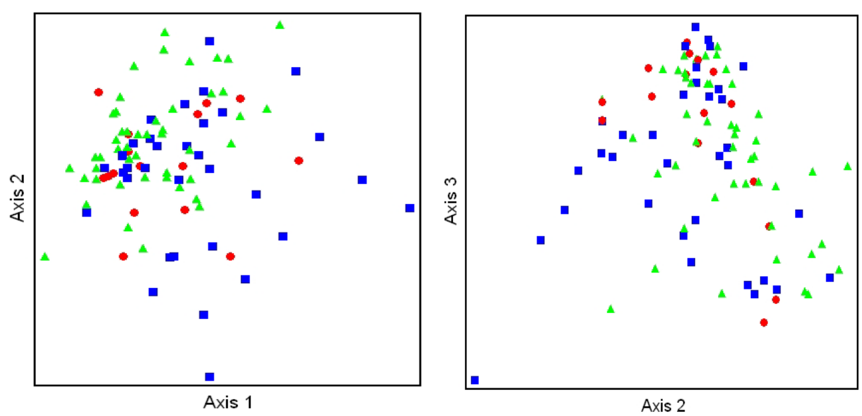


Figure 8. Nonmetric Multidimensional Scaling (NMS) biplots comparing the woody plant community assembly of unburned plots (red circles), plots burned once (green triangles), and plots burned twice (blue squares), in Linville Gorge Wilderness Area, North Carolina. All rare species removed from analysis.

Table 6. Nonmetric multidimensional scaling output showing the most influential woody plant species along each axis in the model.

Axis 1	**r**	**Axis 2**	**r**	**Axis 3**	**r**
Acer rubrum	-0.410	*Kalmia latifolia*	0.554	*Acer rubrum*	0.558
Nyssa sylvatica	0.421	*Vaccinium pallidum*	0.464	*Kalmia latifolia*	-0.537

Species Pearson and Kendall Correlation with Ordination Axes

While there was evidence that burning facilitated woody species, results of the multivariate analyses are less compelling. These communities were highly heterogeneous prior to burning, and they remained so after being burned 1x or 2x. There is evidence, in the form of slightly larger NMS point clouds, that burning resulted in an increase in community heterogeneity. This is not surprising, since the results for Objective 1 and Objective 2 clearly indicate the high degree of variability in fire severity across the landscape, as well as differences in community response to the number and/or severity of burns. The apparent positive relationship between number of burns the relative cover of *Nyssa sylvatica*, along with the negative relationship observed for *Kalmia latifolia*, are supported by observations from the field. *Nyssa sylvatica* was observed to sprout prolifically after fire, although it remained a relatively minor component of the community. High severity fires have been shown to kill *Kalmia latifolia* (Waldrop and Brose 1999). Aside from these changes, however, it is clear that the fires did not substantially alter plant community assembly in these sites.

Management Implications

Both R*d*NBR data from remote sensing and a composite burn index (CBI) that we derived from field data proved to be exceptionally accurate at predicting on-the-ground differences in fire severity. This allowed us to use GIS data to produce an accurate map of fire severity in three landscapes that were burned once and two landscapes that were burned twice. The R*d*NBR method is inexpensive and, if checked by ground truthing, it may prove to be a valuable asset for monitoring burn severity after any type of fire.

Our composite burn index, which was a combination of measurements of soil and vegetation changes after fire, allowed an easy-to-follow system for predicting fire severity throughout all five wildfires. Because it has a high correlation with R*d*NBR, it can also be used to map severity. By following this procedure, we were able to observe the patterns of severity in all 5 burn units, especially because CBI followed similar patterns at all fires. The most severely-burned areas were on the higher elevations, probably ridges, on slopes of moderate steepness, and facing to the south, southwest, west, or northwest. While similar observations have been made during individual fires throughout the Appalachian Mountains, this is the first time that severity has been studied through repeated wildfires in a generalized area. With continued analyses, it may be possible to match burn parameters for prescribed fires with position on landscape for a better prediction of overall impact.

Changes to vegetation after 1 or 2 wildfires followed patterns observed in previous research on prescribed burning. Topographic positions that are typically dry, burned with high severity with the possible exception of mid-slope positions that were steep (>60%). Even with severe burns, species richness and diversity had few changes because of rapid sprouting of hardwood trees and shrubs and germination of pioneer species. Invasive species were rarely observed in study plots suggesting that a seed source was either consumed by fire or was not present.

The effects of 1 burn vs. 2 burns were more pronounced in the dry thermic oak community than in the moist acidic cove community, likely due to the lower severity associated with moist coves. Shrub and tree cover decreased at all levels of re-burn severity. Herb richness and total richness were highest in the highest level of re-burn severity. Significant changes (increases) in species richness occurred only at scales $< 1\ m^2$, which is likely a reflection of large-scale community heterogeneity and/or heterogeneous patterns of burn severity. Woody plant community assembly was not substantially affected by burning, nor did burning appear to promote exotic plants.

In none of our observations was a species eliminated and most were not reduced in cover. The wide variability in topographic positions caused the fires to create widely variable conditions of severity. On dry slopes where trees were killed in large numbers, an early-successional habitat is common and the resulting stand will be even aged. In other areas, overstory trees may have been thinned but will remain the dominant vegetation on the site. If enough of the trees that have been top-killed sprout and survive, the resulting stand will become 2-aged or uneven aged. Unburned areas had some individual tree mortality, which will allow gap-phase replacement, but these stands will likely remain even aged. The combination of wildfire numbers, topographic variability, and preburn vegetation differences will allow the Linville Gorge Area to remain among the most diverse areas in the United States.

Relationship to Recent Findings and Ongoing Work

In the Appalachian Mountains, topography and associated moisture regimes are the primary factors in predicting fire impacts. However, these conditions are highly diverse making prediction of fire behavior and fire effects very complex. Studies of fire behavior and fire effects on vegetation and soil are limited for the southern Appalachian Mountains. Studies of wildfire are rare and may be limited to three done in this same area after one wildfire. Reilly et al. (2006a) found that one fire increased herbaceous species richness at intermediate scales ($\geq$100m^2), but increases in tree species richness were limited to small scales (<10m^2) due to lack of immigration. This finding held true for the current study even after the six years since the last measurement and after a second fire in some areas. As with most studies using prescribed fire, vegetation changes can be minor and short lived (Arthur et al. 1998, Elliott et al. 1999, Kuddes-Fisher and Arthur 2002, and Waldrop et al. 2007). The major changes deal with the loss of overstory trees and changes in ecosystem structure. Tree mortality may take as many as 5 years to observe and it can range from a few trees to all depending on initial tree condition (Yaussy and Waldrop 2010). Where most trees die, plant species richness and diversity do not change in the long term as most woody species sprout and eventually dominate the sprout. Most long-term ecosystem changes are those associated with bird habitat, insect abundance, and predicted wildfire intensity (Waldrop et al. 2014).

After one wildfire, Reilly et al. (2006b) found that severity was correlated with the same environmental gradients structuring spatial patterns of community composition across the landscape and ultimately maintained species turnover and β-diversity. This study takes that finding farther by showing the strong relationship between topographic position and CBI, a measure of burn severity. Wimberly and Reilly (2007) used spatial analysis of remotely sensed images and found that patterns of fire severity were strongly constrained by forest type and topography. The current study did not find a strong relationship with forest type with severity, although cover type is strongly determined by topographic position and the two variables are difficult to separate. These results support the notion that fire promotes landscape heterogeneity, but also suggest that woody regeneration in areas of high tree mortality could be inhibited by strong dispersal limitation in dominant tree species.

Multivariate analyses in this project found a similar pattern in species assemblage ay reported by Phillips et al. (2007). Plant communities tended to be heavily dominated by individual plant species immediately after disturbance, especially if the disturbance killed overstory trees. In that study, heavy tree mortality was the result of mechanical cutting of understory trees and shrubs followed by a hot prescribed fire. In the current study, the same was observed with high intensity wildfire. In both studies, however, dominant pioneer plant species were ephemeral and after 3 to 5 years, plant communities did not vary by different fuel reduction treatments or differences in wildfire severity.

Future Work Needed

This study adds to a body of literature that suggests that changes from single disturbances in hardwood-dominated forests of the southern Appalachian Mountains can be minor and short lived. Even in areas where wildfires burned to produce conditions of high severity, the plants generally survived or regenerated quickly. Therefore, changes in species richness and diversity were not observed several years after the fire. What is not as well understood is the type and number of disturbances that are required to reach management objectives, such as restoration to a particular forest type or to produce desired forest products. Much is known about the autecology of some species but knowledge is scant about the influence of disturbance regimes on previously ignored ecosystem components, such as soils, vertebrates, invertebrates, and pathogens, and their importance to the overall health and productivity of the systems. Fire and other natural disturbances, such as storms, insects, and diseases, may have either positive or negative influences depending on the specific ecosystem component. Anthropogenic disturbances, such as harvesting, herbicides, site preparation and prescribed burning may produce some of the desired effects of natural disturbances but past research has focused on timber production and game habitat with little knowledge of their influence on other ecosystem components.

This study confirmed the value of using a relatively simple spatial analysis to predict fire severity over a relatively large and heterogeneous landscape. Both R*d*NBR and our composite burn index (CBI) were highly correlated with field measurements of fire severity and with each other. Therefore simple GIS data showing the change of each pixel before and after fire gives a good indication of the CBI which, then, serves as a good variable for mapping severity. This procedure allowed us to examine burn severity as it varied across differing topographic variables, thus providing a better understanding of fire impacts over a diverse landscape. While this technique has proven useful twice in the Linville Gorge area, it is yet to be applied to other fires and other disturbances in the southern Appalachian region.

There was no apparent relationship between burn severity and Table Mountain pine regeneration, nor did re-burn severity appear to influence exotic plant invasion. Sufficient research is available to understand that the lack of regeneration of Table Mountain pine could have been caused by consumption of cones, poor seed viability, residual litter and duff layers that were too thick for seedlings to survive, inadequate sunlight or shade for seedlings, or lack of mycorrhizal root tips (Gray et al. 2002, Mohr et al. 2002, Randles et al. 2002, Waldrop et al. 2002). Although a good result, the lack of exotic plants in the study area was a surprise. Descriptions of the Linville vegetation by Newell and Peet (1998) showed the presence of Royal Paulownia (*Paulownia tomentosa*) in sufficient quantities to be a potential invasive after a disturbance such as wildfire. The absence of this and other exotic species after wildfire indicates a lack of understanding of the autecology of exotic and potentially invasive plant species.

Deliverables Crosswalk Table

Proposed	Delivered	Status
Field tour	Vegetative patterns impacts by wildfire frequency and severity. Southern Blue Ridge Fire Learning Network Annual Meeting, Crossnore NC	May 2010
Conference presentation	Reilly, M.; Waldrop, T.; Wimberly, M. 2009. Effects of Initial and Repeated Wildfire in the southern Appalachian Mountains. 7th North American Forest Ecology Workshop; Logan Utah; 22-26 June 2009.	June 2009
Field tours	Current research on fire in the southern Appalachian Mountains. Bent Creek Workshop on oak regeneration. Asheville, NC	July 2011 July 2012 July 2013
Conference presentation	Reilly, Matthew; Waldrop, Thomas; Wimberly, Michael. Effects of initial and repeated wildfire in the southern Appalachian Mountains. AFE 4th International Congress, 2009 Fire Ecology and Management, Savannah, Georgia.	December 2009
Website	Project results will be added to the Wildfire section of the Encyclopedia of Southern Forest Science. http://fire.forestencyclopedia.net/	November 2013
Maps and mechanistic model for fire effects on landscape diversity for display	Reilly, Matthew; Waldrop, Thomas; Wimberly, Michael. 2010. Effects of initial and repeated wildfire in the southern Appalachian Mountains. AFE 4th International Congress, 2009 Fire Ecology and Management, Savannah, Georgia. 30 November- 4 December, 2009.	Maps presented as a part of a poster for the conference. One is in this report and others will be available in the Forest Science Encyclopedia 11/13
Field tour	Waldrop, T.A. Fire on the Mountain – Wildfire vs. Prescribed Fire. What's the Difference? Field trip for the Chattooga River Ranger District, R-8	April 2013
Workshop	Wildland Fire in the Appalachians: Discussions among Managers and Scientists. Oct 8-10, 2013. This work along with other JFSP-funded research will be presented at the workshop hosted by the Consortium of Appalachian Fire Managers and Scientists.	October 2013
Refereed papers	Reilly, M.; Waldrop, T. Assessment of fire severity and species diversity in the southern Appalachians using Landsat TM and ETM+ imagery. Remote Sensing of Environment. Hagan, D.; Waldrop, T.A. Impacts of repeated wildfire on vegetation in the southern Appalachian Mountains. International J. Wildland Fire.	Drafts included as one paper in appendix of this report. Both will be revised and submitted by February 2014.

Appendix 1

DRAFT MANUSCRIPT

Abstract-- The infrequent occurrence of large wildfires in the southern Appalachian Mountains over the last several decades has offered few opportunities to study the impacts of these types of disturbances. As a result, relatively little is known about how heterogeneity in topography, vegetation, and recent disturbance history interact to influence patterns of fire severity across the landscape. Since 2000, five separate wildfires burned a large portion of the area in, and surrounding the Linville Gorge Wilderness in western North Carolina, two burned the same area a second time. Burn severity and vegetative recovery were measured in 152 plots established in 1992 prior to any wildfires. Field measurements of severity were strongly related to GIS data, allowing an accurate map of severity conditions in all 5 burned landscapes. Severity followed similar patterns at all fires. The most severely-burned areas were on the higher elevations, probably ridges, on slopes of moderate steepness, and facing to the south, southwest, west, or northwest. The effects of 1 burn vs. 2 burns were more pronounced in dry-site than in moist-site communities. Shrub and tree cover decreased at all levels of re-burn severity. Herb richness and total richness were highest in the highest level of re-burn severity. Significant changes (increases) in species richness occurred only at scales < 1 m^2, which is likely a reflection of large-scale community heterogeneity and/or heterogeneous patterns of burn severity. Woody plant community assembly was not substantially affected by burning, nor did burning appear to promote exotic plant invasion. Our measurements and maps of fire severity will help managers to better understand vegetative response to wildfire across widely diverse topographic and severity conditions.

Introduction

The acreage of land burned annually in the Appalachian region has increased steadily over the last thirty years with the rate of increase nearly doubling in recent years (Lafon et al. 2005). Projected increases in fire activity in the southeast due to climate change strongly suggest the area of recently burned land in the southern Appalachian Mountains is likely to continue increasing (Bachelet et al. 2001). As a result there will be a growing need for managers to understand the effects of fires returning to recently burned stands in relation to how repeated fire affects management goals. Information on re-burns in the southern Appalachian Mountains is limited (Arthur et al. 1998) and managers in the region can only use inference from past studies of re-burns in the western US for insights into effects of previous fires on re-burn severity (Thompson et al. 2007, Romme 1982). With little information available from within the region, managers of forests in the southern Appalachian Mountains have a need to understand the effects of repeated fire on woody regeneration, understory diversity, and invasion of exotic species.

Fire historically increased landscape heterogeneity in the southern Appalachian Mountains by maintaining a variety of community types across the landscape (Delcourt and Delcourt 1998). Anthropogenic fires perpetuated oak and chestnut (*Castanea dentata*) on upper slopes and fire adapted species such as Table Mountain pine and pitch pine (*Pinus rigida*) on drier ridges. A long period of fire exclusion has left many forest communities in a degraded state due to encroachment of more fire sensitive species (Williams 1998). Active prescribed fire programs began in the region as late as the 1980's and have gradually increased in importance since (Waldrop et al. 2007).

Although some research has been done on the effects of fire in the southern Appalachian Mountains, the information available to managers has been largely derived from the results of single relatively small prescribed fires intended to restore decadent Table Mountain pine (*Pinus pungens*) stands, promote oak (*Quercus* spp.) regeneration, and increase understory diversity and productivity (Turrill-Welch and Waldrop 2001, Kuddes-Fisher and Arthur 2002, Elliott et al. 1999) The rare occurrence of large wildfires in the region (Lafon et al. 2005) has offered few opportunities to study the effects of this type of disturbance at the landscape scale. In the fall of 2000 a wildfire burned the majority of Linville Gorge in western North Carolina after a prolonged drought and infestation of southern pine beetle. The existence of an extensive network of vegetation plots established prior to the 2000 fire enabled researchers to study how wildfire maintains both community and landscape level diversity (Reilly et al. 2006a, Reilly et al. 2006b), as well as the influences of topography and forest type on spatial patterns of fire severity (Wimberly and Reilly 2007). In the spring of 2007 two separate wildfires burned a portion of the landscape that burned in 2000 as well as much of the remaining unburned area within and surrounding Linville Gorge. This rare event was followed by another wildfire in 2008 that burned previously unburned areas of the Gorge. The occurrence of these fires provided a unique opportunity to follow up on previous work by re-measuring an extensive network of plots to examine the impacts of multiple disturbances on a number of biological variables of interest to land managers.

Studies of prescribed fire have documented effects on vegetation structure, woody regeneration of pine and oak species, and diversity in the understory (Turrill-Welch and Waldrop 2001, Kuddes-Fisher and Arthur 2002, Elliott et al. 1999, Waldrop et al. 2007). Table Mountain pine, a fire dependent species endemic to the southern Appalachians, has been of particular interest (Waldrop and Brose 1999, Welch et al. 2000, Turrill-Welch and Waldrop 2001). Studies on this species have elucidated microsite conditions and levels of fire severity necessary for seedling establishment. Effects of repeated fire in regenerating Table Mountain pine stands are unknown. In oak dominated forests, fire is thought to promote regeneration by reducing the density of undesirable hardwood saplings and midstory shrubs and increasing light levels (Van Lear and Watt 1993). However, studies on the application of single prescribed fires in oak forests found little effect on oak regeneration and species composition in the understory due to vigorous sprouting of other hardwood species and shrubs and suggest that repeated burning or fires of higher intensity may be necessary (Kuddes-Fisher and Arthur 2002).

A rare study of large wildfires in the southern Appalachian Mountains was done in Linville Gorge after a wildfire in 2000. Fire increased herbaceous species richness at intermediate scales ($\geq 100m^2$), but increases in tree species richness were limited to small scales ($<10m^2$) due to lack of immigration (Reilly et al. 2006a). Severity was correlated with the same environmental gradients structuring spatial patterns of community composition across the landscape and ultimately maintained species turnover and β-diversity (Reilly et al. 2006b). Spatial analysis of remotely sensed images found that patterns of fire severity were strongly constrained by forest type and topography (Wimberly and Reilly 2007). These results support the notion that fire promotes landscape heterogeneity, but also suggest that woody regeneration in areas of high tree mortality could be inhibited by strong dispersal limitation in dominant tree species.

Current knowledge of fire effects suggests two possible outcomes dependent on fire severity in re-burned areas. Information on re-burns is currently limited (Arthur et al. 1998) but past studies on areas subject to re-burn in the western US provide some insight on potential burn severity and effects in Linville Gorge. Despite contrasting results, these studies agree that fire severity depends on the potential of young regeneration to carry fire (Thompson et al. 2007, Romme 1982). The abundant pine regeneration and vigorous sprouting of hardwood trees and ericaceous shrubs typical after fire in the southern Appalachians suggest that fuels are likely capable of carrying severe fire. Re-burn severity in recently regenerated conifer stands in Oregon was higher than that of the initial burn (Thompson et al. 2007). If this is the outcome of the 2007 and 2008 fires and young Table Mountain pine regeneration is destroyed without having produced sufficient cones, subsequent regeneration will depend on input of seeds from outside areas. Likewise, if oak trees are killed by higher severity fire in the re-burned areas, regeneration will also depend on immigration of acorns. These outcomes could ultimately affect landscape diversity and productivity. However, if Table Mountain pines have sufficient seed stock and mature oaks survive; higher severity re-burns may be more successful than initial burns at eliminating understory shrubs and saplings and reducing litter and duff depth. These effects would further promote regeneration of Table Mountain pine and desired hardwoods as well as promoting diversity in the understory. Mountain laurel grows in dense thickets and represents a major source of vertical fuels, occasionally supporting crown fires (Waldrop and Brose 1999). Control of this species would reduce wildfire hazard and promote regeneration of desirable species.

This project was conducted to determine how fire severity and fire effects differ among areas with different recent burn histories. Specific objectives included:

1. To compare fire effects and spatial patterns of severity between areas measured before burning and after one (2000 and 2008) or two (2007 or 2008) wildfires,
2. To assess the effects of re-burn severity on woody regeneration (Table Mountain pine and oak species), understory species diversity, and exotic species invasion, and
3. To examine successional change in areas with different recent disturbance histories and effects on species turnover and diversity across the landscape.

Methods

Study Site

Linville Gorge is in the Pisgah National Forest south of Boone, NC and contains a 4390 ha federally designated wilderness area. Elevations range from 320 m at the bottom of the gorge to 1250 m on the upper slopes of the ridges. Topography is complex and the presence of steep environmental gradients has resulted in an extremely diverse landscape, much of which has never been logged and is thought to be representative of pre-settlement forests (Newell and Peet 1998). Upper slopes and ridges are dominated by thermic pine and oak communities with a thick layer of ericaceous shrubs. Thermic pine forests are composed predominantly of Table Mountain pine, pitch pine, and Virginia pine (*Pinus virginiana*). Thermic oak forests are dominated by chestnut oak (*Quercus prinus*), scarlet oak (*Quercus coccinea*), and white oak

(*Quercus alba*). Mountain laurel, a major component of live fuel, is abundant in both. Lower slopes and coves are dominated by chestnut oak, white pine (*Pinus strobus*) and eastern hemlock (*Tsuga canadensis*) with a thick layer of rhododendron (*Rhododendron* spp.) in the understory. Prior to 2000, the last widespread surface fires occurred in the early 1950's. In 2000 a wildfire burned approximately 4000 ha in and surrounding Linville Gorge (Table 1). Fire severity was heterogeneous across the landscape with high severity crown fires occurring on steep slopes and upper ridges and low severity surface fires occurring on mid-slopes and coves (Fig. 1a) (Wimberly and Reilly 2007). In spring 2007, two separate fires burned a large portion of the landscape previously burned in 2000 as well as much of the remaining unburned area surrounding Linville Gorge (Fig 1b). Another large wildfire occurred in 2008 and burned much of the area immediately adjacent the area that burned in 2000 and 2007.

Table 1. Area impacted by 0, 1, or 2 wildfires.

Fire History	Area (ha)
Unburned	2,031
Burned once in 2000	1,632
Burned once in 2007 or 2008	1,602
Burned twice, 2000 and 2007	2,477

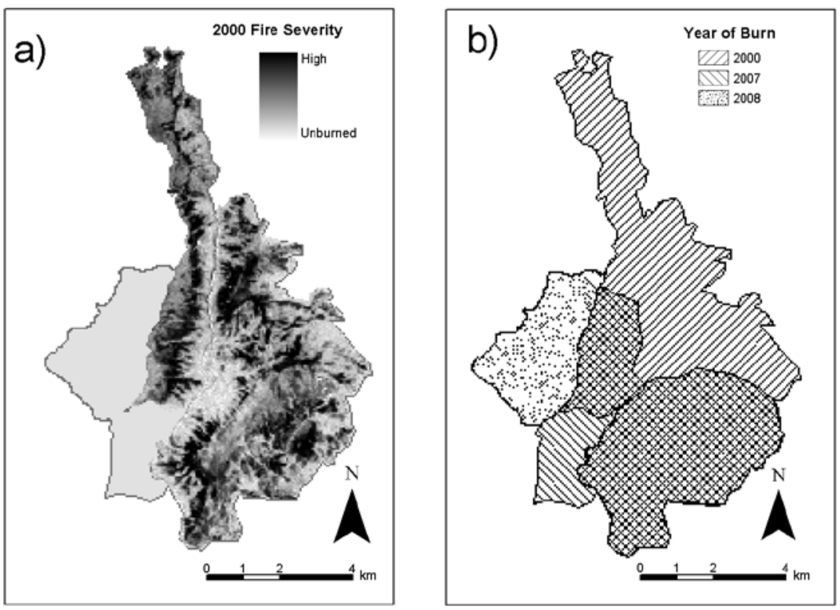


Figure 1. Linville Gorge maps showing a) fire severity in 2000 (from Wimberly and Reilly 2007) and b) approximate boundaries of the 2000, 2007, and 2008 wildfires.

Sampling

Plots were characterized as burned twice, burned once in 2000, burned once in 2007 or 2008, and unburned. We re-measured two previously established sets of plots and established new plots to increase sample size and spatial coverage in underrepresented areas. The effects of fire on plant community assembly were assessed using Analysis of Variance (ANOVA) and Nonmetric Multidimensional Scaling (NMS).

The first set of previously-measured plots consists of a series of 181 intensively sampled preburn vegetation plots established in 1992 to document the variation in composition and structure of vegetation in Linville Gorge (Newell and Peet 1998). Plots were stratified to capture topographic and geologic variation and were sampled following the North Carolina Vegetation Survey (NCVS) protocols (Peet et al. 1998). This sampling methodology involved intensive sampling of the understory in a series of nested subplots ranging from 0.01 m^2 to 10 m^2 within one to four adjacent 100 m^2 modules. All species were counted and assigned cover estimates at the 100 m^2 scale. Woody vegetation $\geq$1.4 m diameter at breast height was sampled within each 100 m^2 module.

The second set of plots was a series of 57 rapid assessment plots established in 2003 to capture the spatial variation of fire severity across the landscape. A modified version of Key and Benson (2002) was used to create a Composite Burn Index (CBI). This was based on a visual estimate of the percent mortality of dominant canopy trees and mid-canopy trees within a 30 m radius and correlated strongly with changes in basal area (Wimberly and Reilly 2007). Of these plots, 49 burned once in 2000 and 8 burned twice, 2000 and 2007. All rapid assessment plots were measured after the 2000 wildfire.

We re-measured the majority of NCVS plots using the same measurements that were taken in 1992. Plot re-measurement was determined by the number of plot markers that were found and the patterns of fires that burned through the plots. Our goal was to measure 50 or more plots that had been burned once, twice, or not at all (Table 2). Special emphasis was given to find and measure those plots that were measured both before and after the 2000 wildfire. Field crews spent approximately 8,500 person hours locating and measuring the NCVS plots over the growing seasons of 2009, 2010, and 2011. Decay of reference trees, inaccurate GPS points from old equipment, and loss of monuments since 1992 contributed to the loss of many of the original plots. However, we exceeded our goal of finding and measuring 150 plots with a total of 154. Of those, 53 were unburned, 51 were burned once, and 50 were burned twice.

Table 2. Number of plots previously measured and re-measured by fire history and sample type.

Plot Type and Burn History	Measured in 1992 (pre-burn)	Measured in 2003 (after first wildfire)	Re-measurement
NCVS Intensive Sample Plots			
Unburned	58	11	53
Burned once in 2000	45	14	14
Burned once in 2007 or 2008	23	0	37
Burned twice, 2000 and 2007	55	21	50
Rapid Assessment Plots			
Unburned	0	0	0
Burned once in 2000	0	49	49
Burned once in 2008	0	0	26
Burned twice, 2000 and 2007	0	8	8

Additional response variables were measured on NCVS plots at the module scale (100 m^2) and in three 1 m^2 subplots following protocols of Waldrop and Brose (1999). Severity in subplots was estimated using the following categories: 1) unburned, 2) burned with partially consumed litter present, 3) no litter present and 100% of the area covered by duff, 4) soil exposure on 1-30% of the area, 5) soil exposure on 31-60% of the area, and 6) soil exposure on 61-100% of the area. Insolation was estimated between 10:00 hours and 14:00 hours on sunny days and described as one of the following categories: 1) Full shade, 2) 1-30% of the area receiving direct sunlight, 3) 31-60% of the area receiving direct sunlight, and 4) 61-100% of the area receiving direct sunlight. Density of all seedlings and hardwood sprouts were counted in each 1 m^2 subplot by species. Litter and duff depths were taken at one corner of each subplot. At the module level, severity and insolation was estimated again, as well as cover of litter, duff, mineral soil, rock, and downed wood.

All 57 of the rapid assessment plots were re-measured using the same procedures as before. An additional 26 plots were established in the area burned once in 2008. Multi-temporal Landsat Imagery was used to investigate spatial patterns of fire severity across the landscape. Fire severity was estimated using the difference in the Normalized Burn Index (*d*NBR) between pre- and post-fire imagery. Spatial data for *d*NBR from the 2000 fire already exist, so only the pre- and post-fire imagery had to be acquired from the USGS for the 2007 and 2008 wildfires.

Data Analysis

<u>Objective 1</u>- Comparing initial fire effects and spatial patterns of severity between areas subject to initial burn and re-burn in the 2007 or 2008 fires.

Initial fire effects were compared using an ANOVA design. Comparing fire severity in re-burned areas to those burned once followed Thompson et al. (2007). This method used the *d*NBR from pre- and post-fire Landsat imagery and a series of random points which were incorporated into generalized least squares regression models. Assessing the roles of topography and forest type between the two areas followed Wimberly and Reilly (2007). Accuracy of the *d*NBR values was estimated using CBI values calculated from field plots using a simultaneous autoregressive nonlinear regression model that explicitly incorporates spatial autocorrelation. This model was used to create a spatially explicit map of predicted CBI values to be used with a digital elevation model and detailed vegetation map (Newell and Peet 1998) to compare the roles of topographic variables and forest type in determining fire severity between the initial fire and re-burn.

We obtained pre-and post-fire Landsat imagery to calculate the Relativized difference in the Normalized Burn Ratio (R*d*NBR) (Miller and Thode 2007) for each of the wildfires in 2007 and 2008. The best available images used to calculate the R*d*NBR from the 2007 fires were from 6/10/2006 and 8/16/2007. The best available images used to calculate R*d*NBR for the 2008 fire were from 8/16/2007 and 9/3/2008. We created a standardized composite burn index (CBI) to calibrate the R*d*NBR values from the 2007 fires and the 2008 fire as opposed to comparing them directly due the lack of availability of imagery on anniversary dates and the resulting differences in time since fire. We attempted to remove the bias due to different time since fire by using different models for each year to predict an index that has been calibrated to actual fire effects.

We used a modified version of the Fire Effects Inventory and Monitoring (FIREMON) protocol (Key and Benson 2002) to visually estimate fire effects in four categories within a 15 meter radius of plot center on a 300 point scale. The four categories included % duff consumption, % midstory and shrub topkill, % mortality of midstory trees, and % mortality of overstory trees. These particular metrics were chosen because they were all measured consistently across the various sources of field data, and because they could be reliably measured 3–4 years after the fire. These metrics were recorded as index values ranging between 0 and 3, with higher numbers indicating greater mortality (Key & Benson, 2002), and the CBI was a computed as a weighted average with weights of 1, 2, and 3 assigned to the shrub/small tree, intermediate tree, and large tree strata. This weighting put more emphasis on the mortality of larger trees, which were the most reliable field-based indicators of long-term fire effects. The composite CBI values were rescaled to range between 0 and 1, with unburned plots assigned a CBI value of zero. Each plot was associated with the dNBR measurement from the nearest pixel by overlaying plot centroids on the dNBR image. Non-linear regression was used to fit an equation relating the observed CBI indices to dNBR, and this equation was then used to predict CBI for all pixels falling inside the fire perimeter. All pixels in each wildfire were then sampled for CBI, elevation, topographic moisture, slope, and aspect. We compared the distributions of CBI values across each of these gradients.

<u>Objective 2</u>- Assessing the effects of re-burn severity on woody regeneration, understory species diversity (richness and species turnover), and exotic species invasion.

Data analysis for Objective 2 was similar to Reilly et al. (2006a) but required a repeated measures approach instead of a paired approach. Pre-fire data at the species level was limited to cover at the module scale, but density of woody regeneration was estimated from subplots to allow an assessment of how well cover predicts density. The NCVS sampling methodology allows examination of species richness at multiple spatial scales within a plot (Peet et al. 1998). Species-area curves were created using non-linear regression and coefficients (small scale richness) and slopes (turnover within the plot) were compared for changes. These were correlated with fire severity as estimated from changes in basal area to assess the role of these processes and compare their relative importance after re-burn to that of initial burn. The six severity classes measured in the field were combined to three severity classes: low (unburned to burned with partially consumed litter present), medium (no litter present and 100% of the area covered by duff to soil exposure on 1-30% of the area) and high (soil exposure on 31-100% of the area). Plots were described by community types and repeated measures ANOVA was used to test for changes among the three sampling periods in all response variables. Since only a fraction of study plots burned twice, data for thermic oak and acidic cove communities were pooled.

<u>Objective 3</u>- Examining successional change in areas with different recent disturbance histories and compare species turnover and diversity across the landscape from 1992 to 2009.

Analyses for this objective followed those used in Reilly et al. (2006b). Nonmetric Multidimensional Scaling was used to ordinate plots and to determine compositional similarity.

Results

Objective 1

All wildfires resulted in extremely heterogeneous patterns of fire severity across the landscape ranging from low levels of litter consumption to total litter and duff consumption and complete midstory and canopy mortality. R*d*NBR values were strongly correlated with field based estimates of fire effects as well as CBI values (Table 3). When possible, we compared results among 5 different burn units: Dobson Knob (burned once), Shortoff Mountain (burned twice), Pinnacle (burned once), Pinnacle (burned twice), and Sunrise (burned once) (Fig. 2).

Table 3. Relationship of R*d*NBR values to field measure of fire severity and the Composite Burn Index.

Variable	R^2 for plots burned in 2007	R^2 for plots burned in 2008
% Duff Consumption	0.55	0.60
% Midstory and Shrub Topkill	0.62	0.69
% Mortality of Subcanopy Trees	0.50	0.55
% Mortality of Canopy Trees	0.70	0.73
CBI	0.76	0.78
CBI Model	y=1.58x+224	y=1.33x+156

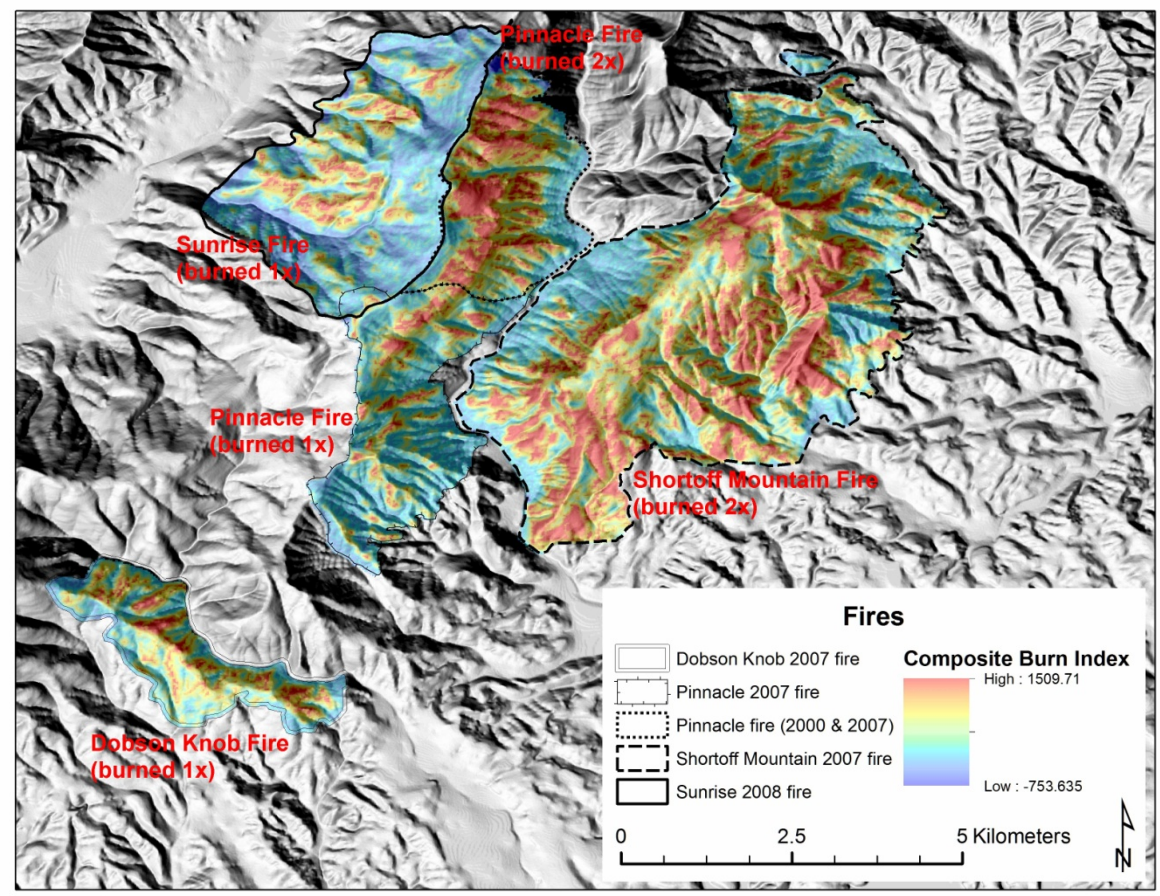


Fig. 2. Predicted Composite Burn Index (CBI) values for the 2007 and 2008 wildfires in and around Linville Gorge.

The distributions of CBI values over major environmental gradients were similar among each of the five landscapes (Fig. 2), although they were highly variable within each landscape. CBI values were generally higher in landscapes subject to repeated fire, particularly the Shortoff Mountain Fire. CBI values within the area burned twice in the Pinnacle Fire were higher than those in burned once in the Pinnacle Fire and Sunrise Fire. CBI values within the area burned once in the Dobson Knob Fire area were similar to those burned twice in the Pinnacle Fire except over the elevational gradient where those burned twice in the Pinnacle Fire were higher.

CBI values tended to be highest at the highest elevations where ridges burned severely (3,000 to 3,500 feet) but there were no statistical differences (Fig 3a). Fires were less severe in areas with higher TMI values but there were no significant differences. TMI values were over 800 if TMI was less than 1, but were approximately 700 if TMI was greater than 1 (Fig. 3b). Severity was greatest on moderate slopes as compared to flatter or steeper slopes (Fig 3c). CBI was approximately 800 at slopes of 30 to 60 percent but only about 650 on slopes at 15-30 percent and 60-75 percent. Aspect was most closely related to post-burn severity (Fig. 3d). Although there was high variability, CBI was significantly lower on north-facing slopes (425) than on west- and southwest-facing slopes (975). Other slopes that had conditions of high severity included those facing to the south and northwest.

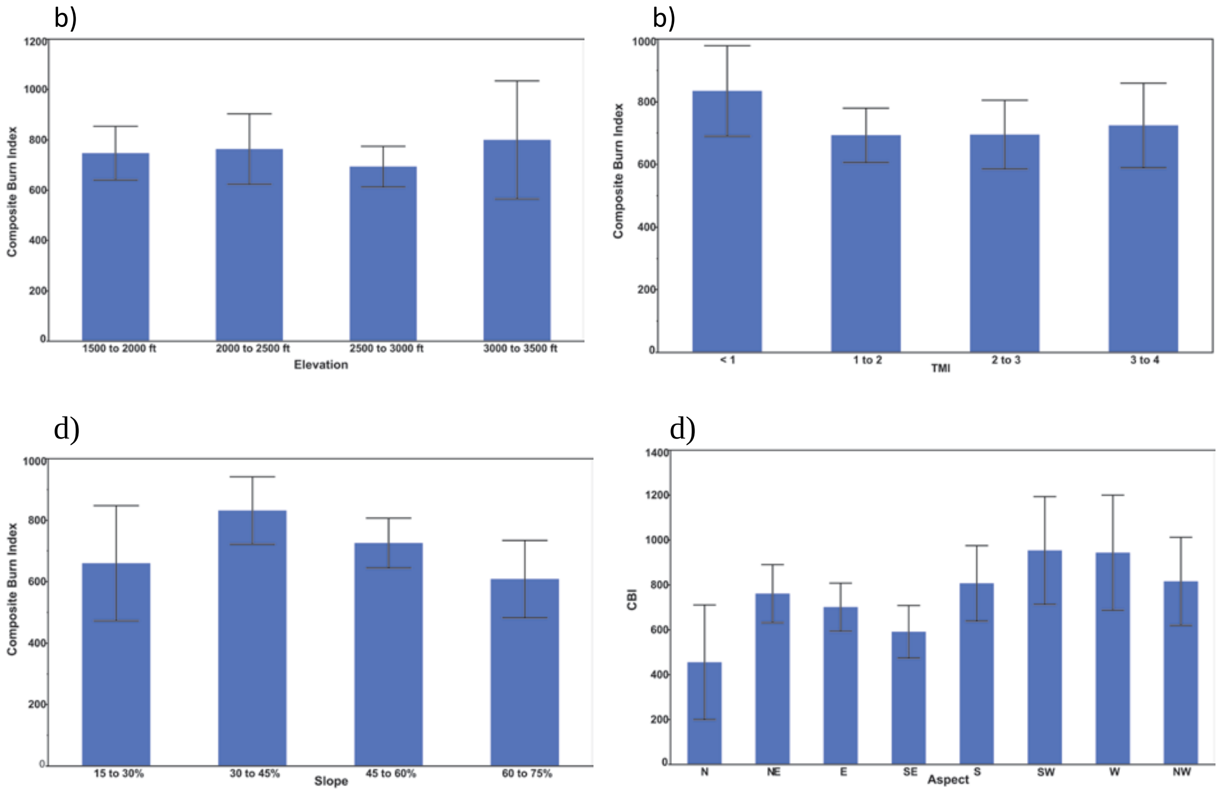


Fig. 3. The distribution of CBI values across a) elevation, b) topographic moisture, c) slope, and d) aspect for the 2007 and 2008 wildfires.

The strong relationship of fire severity to topography provided a problem for analysis because topography had to be included. However, the highly diverse topography of Linville Gorge provided an almost infinite number of site and cover types. Therefore, we chose to focus our analyses of plant response to two site type at either extreme of the moisture gradient, moist acidic coves and dry thermic oak sites. Acidic cove communities occupied lower slopes below bluffs and were dominated by chestnut oak (*Quercus prinus*), red maple (*Acer rubrum*), eastern white pine (*Pinus strobus*), Carolina hemlock (*Tsuga canadensis*), great rhododendron (*Rhododendron maximum*), and associated species. Thermic oak forests occupied upper slopes or ridges and were dominated by chestnut oak, white oak (*Quercus alba*), scarlet oak (*Quercus coccinea*), Eastern hemlock (*T. Canadensis*) and various pine species.

At the plot level, the effects of fire on plant cover varied by community type (acidic cove or thermic oak), and were – in some cases – influenced by the number of burns. For acidic cove communities, decreases in total plant cover were only observed in unburned plots (Table 4). While total plant cover was not influenced by fire in acidic cove plots, the relative contributions of some growth forms did change after burning. Graminoid and herb cover, for example, increased approximately five-fold in plots burned twice. Herb cover increased more than four-fold in plots burned once. No such effects were observed for shrubs, trees and vines in acidic cove plots. In the thermic oak community, significant decreases in total cover occurred in all burn categories. In plots burned twice, there was an eleven-fold increase in graminoid cover and large decreases in tree and shrub cover. Decreases in tree cover also occurred in plots burned once.

Burning also influenced species richness (Table 4). In acidic cove plots, nearly two-fold increases in total species richness were observed in plots burned once and twice, largely due to substantial increases in shrub, herb and tree richness. Similar patterns were observed in the thermic oak community, although the effect of one burn versus two burns was not evident. Species richness increased by over 40% in plots burned once and nearly 74% in plots burned twice. These increases corresponded with increases in the richness of graminoids, herbs, trees and vines. Shrubs were the only growth form for which burning appeared to have no influence on species richness.

Largely in concert with the above effects on cover and richness, burning also influenced diversity indices (Table 4). For the acidic cove community type, the effects were more pronounced for the Shannon index than they were for the Simpson’s index. Significant increases in Shannon diversity for all species combined were observed in plots burned both once and twice. Graminoid diversity (Shannon and Simpson) increased in plots burned twice and Shannon diversity for shrubs increased in plots burned once. For the thermic oak community type, significant increases in Shannon and Simpson diversity occurred for all species combined, graminoids, herbs and shrubs. Significant increases in Shannon and Simpson diversity for vines were observed in plots burned once.

Table 4. Before and after comparisons of cover, species richness, Shannon Diversity and Simpson's diversity (by community type and growth form) in unburned sites and sites burned 1x or 2x. Before and after pairs in bold and highlighted are statistically different at alpha = 0.05.

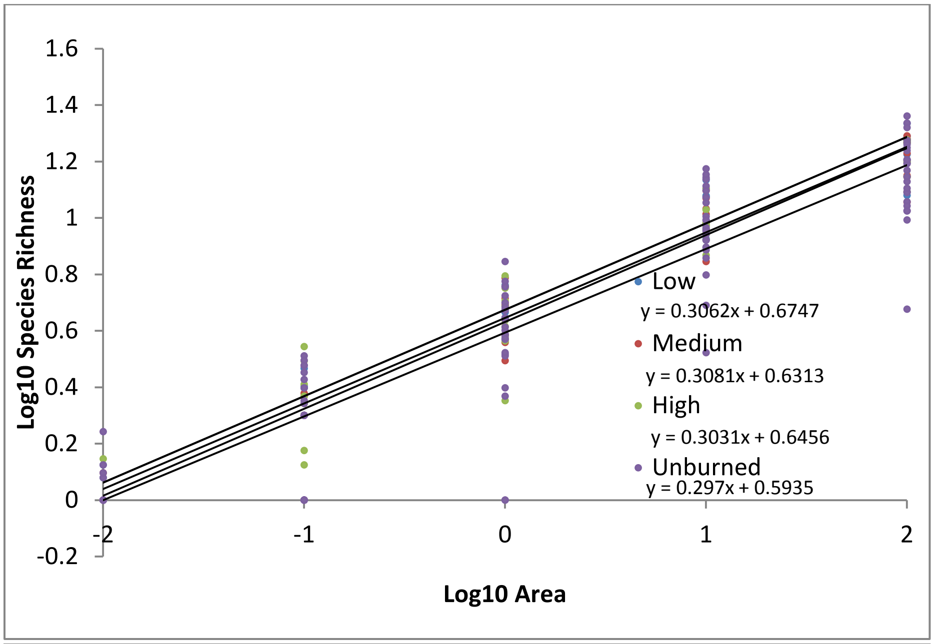
			Cover		Richness		Shannon		Simpson	
			Before	After	Before	After	Before	After	Before	After
Acidic Cove	All species	Unburned	**3.415**	**2.371**	16.538	17.077	1.679	1.692	0.730	0.713
		Burned 1x	2.443	2.029	**18.455**	**31.182**	**1.795**	**2.405**	0.755	0.830
		Burned 2x	2.172	1.541	**24.57**	**48.001**	**2.092**	**2.820**	0.816	0.847
	Graminoids	Unburned	0.001	0.002	0.154	0.154	0.000	0.000	0.000	0.000
		Burned 1x	0.008	0.024	1.091	1.727	0.232	0.420	0.135	0.242
		Burned 2x	**0.024**	**0.114**	**1.142**	**5.000**	**0.128**	**1.161**	**0.061**	**0.590**
	Herbs	Unburned	0.009	0.035	5.615	5.846	1.324	1.361	0.611	0.630
		Burned 1x	**0.020**	**0.082**	7.091	12.364	1.286	1.999	0.570	0.769
		Burned 2x	**0.017**	**0.119**	**11.429**	**22.429**	1.860	2.507	0.764	0.860
	Shrubs	Unburned	1.261	1.034	3.692	4.308	0.463	0.443	0.254	0.223
		Burned 1x	0.671	0.526	**3.455**	**7.091**	**0.594**	**1.196**	0.322	0.559
		Burned 2x	0.597	0.425	**3.857**	**7.286**	0.655	1.261	0.356	0.556
	Trees	Unburned	2.123	1.300	6.769	6.769	1.477	1.301	0.725	0.658
		Burned 1x	1.744	1.397	**6.818**	**10.000**	1.382	1.627	0.679	0.714
		Burned 2x	1.534	0.883	**8.142**	**13.286**	1.582	1.879	0.745	0.786
	Vines	Unburned	0.021	0.000	0.308	0.000	0.067	0.000	0.037	0.000
		Burned 1x	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
		Burned 2x	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Thermic Oak	All species	Unburned	**3.721**	**2.781**	34.154	30.231	0.679	0.996	0.302	0.465
		Burned 1x	**4.017**	**2.175**	**26**	**38.571**	0.681	1.294	0.328	0.500
		Burned 2x	**2.658**	**1.332**	**29.83**	**49.347**	**0.981**	**2.569**	**0.447**	**0.834**
	Graminoids	Unburned	0.001	0.001	0.308	0.769	0.055	0.035	0.033	0.025
		Burned 1x	0.000	0.012	**0.143**	**1.571**	0.000	0.406	0.000	0.232
		Burned 2x	**0.010**	**0.110**	**0.450**	**1.965**	**0.303**	**1.298**	**0.181**	**0.603**
	Herbs	Unburned	0.103	0.058	9.692	8.462	0.874	1.201	0.387	0.541
		Burned 1x	0.107	0.110	**5.500**	**10.643**	0.752	1.364	0.356	0.537
		Burned 2x	0.071	0.095	**6.571**	**20.286**	**1.010**	**2.481**	**0.465**	**0.836**
	Shrubs	Unburned	1.121	0.960	9.923	7.077	1.138	0.969	0.532	0.463
		Burned 1x	1.194	0.725	8.429	9.929	1.230	1.456	0.577	0.610
		Burned 2x	**0.856**	**0.257**	7.952	8.810	**1.264**	**1.665**	**0.584**	**0.718**
	Trees	Unburned	2.303	1.646	12.462	12.000	1.847	1.776	0.783	0.767
		Burned 1x	**2.602**	**1.141**	**10.357**	**14.214**	1.793	1.755	0.774	0.730
		Burned 2x	**1.560**	**0.673**	**12.714**	**15.619**	1.957	1.920	0.812	0.773
	Vines	Unburned	0.193	0.116	1.769	1.923	0.463	0.547	0.322	0.384
		Burned 1x	0.114	0.187	**1.571**	**2.214**	**0.298**	**0.668**	**0.197**	**0.434**
		Burned 2x	0.161	0.197	**2.143**	**2.667**	0.520	0.735	0.350	0.03

Species richness increased with increasing burn severity, but the significance of these effects was dependent on scale (Table 5). At the larger spatial scales (100 and 10 m^2) increases in richness – while substantial – were not statistically significant. At smaller spatial scales (1, 0.1 and 0.01 m^2), however, species richness in plots re-burned at high severity was significantly higher (1.21) than in those re-burned at low severity (0.04).

Table 5: Change in species richness at various spatial scales for plots re-burned at different severities in Linville Gorge Wilderness Area, North Carolina. Values for a given scale with different lower case letters are statistically different at alpha = 0.05.

	Change in species richness					
Scale (m^2)	**Low severity**		**Medium severity**		**High severity**	
100	15.11	a	23.67	a	25.61	a
10	7.58	a	11.12	a	14.61	a
1	2.56	b	4.62	ab	6.77	a
0.1	0.22	b	1.63	ab	2.21	a
0.01	0.04	b	0.71	ab	1.21	a

Comparisons of species area curves shed additional light on the relationship between re-burn severity and spatial scale. The slopes and intercepts of log-transformed curves were similar prior to the initial burn (Fig. 4). After the second burn, differentiation among the lines becomes readily apparent. All re-burned plots had similar slopes, but they had different intercepts. Plots burned by high-severity fires had the highest intercept, followed by those burned at medium and low severities. Divergence in richness from unburned plots increased with increasing spatial scale.



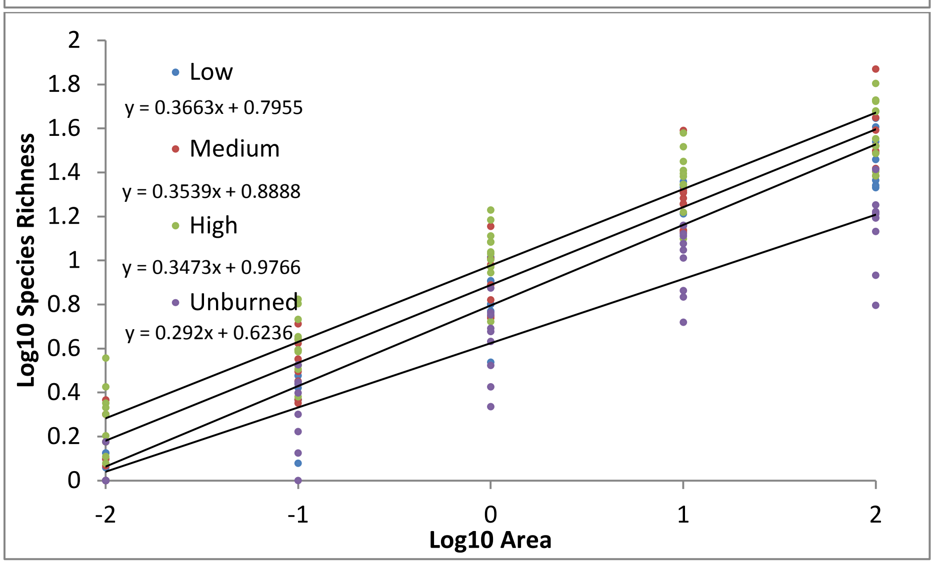
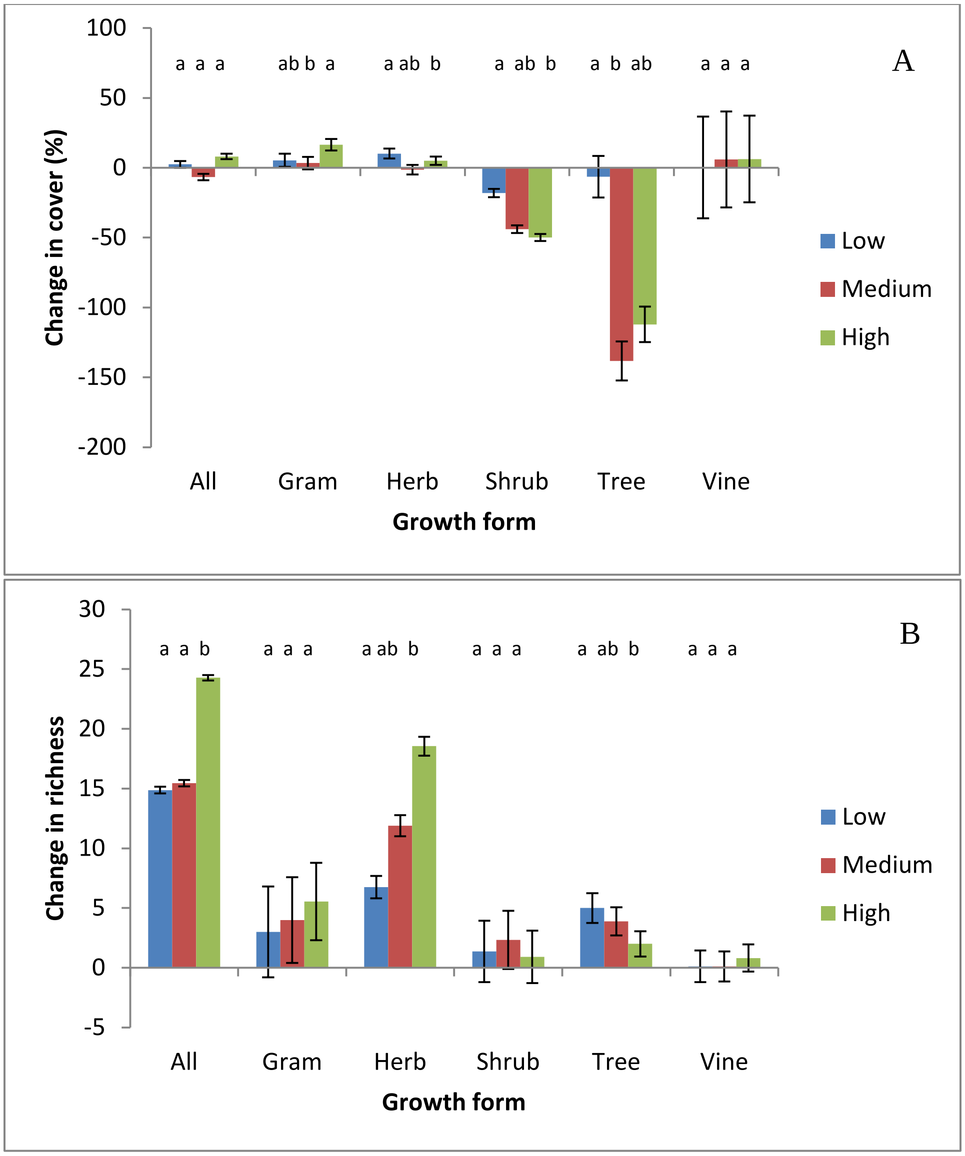


Figure 4. Log-transformed species area curves for plots re-burned at different degrees of severity in Linville Gorge Wilderness Area, North Carolina (Pre-fire: top, Post-fires bottom).

Total cover plant for all 3 burn severity categories did not change substantially following the second fire. However, substantial changes did occur for certain growth form types. Plots re-burned at medium or high severity, for example, saw more than double the decrease in shrub cover than those re-burned at low severity. The decrease in tree cover was even more dramatic. Graminoid cover increased the most in plots subjected to a high severity second burn, while herb cover increased most in the low severity re-burn plots. Vine cover increased significantly with increasing re-burn severity (Fig 5A). Following the second fire, total cover and tree cover were highest in low-severity reburn plots, although differences with other burn severity categories were not statistically significant. Post-fire graminoid and vine cover were highest in the highest burn severity class, and herb cover was highest in the lowest burn severity class. These differences were statistically significant (Fig. 6A).

Species richness increased at all levels of re-burn severity. However, the most substantial (and statistically significant) increases in total species richness occurred in plots subjected to a high severity second burn. These plots experienced an increase of approximately 10 more species than those burned at medium or low severity. For herbs, the magnitude of change increased significantly with increasing re-burn severity (Fig. 5B). Post-fire total species richness was highest in the highest re-burn severity category. This difference was primarily due to the high post-fire cover of herbs in these plots (Fig. 6B).

Re-burn severity also influenced species diversity. Increases in Shannon diversity were observed for all species combined, as well as for each of the five growth form types. The largest increases in Shannon diversity occurred for herbs, and these increases were more pronounced in plots re-burned at either medium or high severity. After the second burn, Shannon diversity for all species combined was highest in plots burned at high severity and this pattern held true for all growth forms except for graminoids. Herbs, however, were the only growth form for which a significant difference between burn categories was observed. Similar patterns were observed for the Simpson diversity index, with medium and high severity fires generally resulting in larger increases in diversity than low intensity fires. Graminoids were the only exception. Substantial increases for this growth form type were observed in all burn severity categories (Fig. 5C and 5D). Post-fire, Simpson diversity for all species combined was highest in medium and high intensity re-burn plots, and this pattern held true for most growth forms (Figs. 6C and 6D).


A
Change in cover (%)
100
50
0
-50
-100
-150
-200
a a a
ab b a
a ab b
a ab b
a b ab
a a a
All
Gram
Herb
Shrub
Tree
Vine
Growth form
Low
Medium
High
B
Change in richness
30
25
20
15
10
5
0
-5
a a b
a a a
a ab b
a a a
a ab b
a a a
All
Gram
Herb
Shrub
Tree
Vine
Growth form
Low
Medium
High

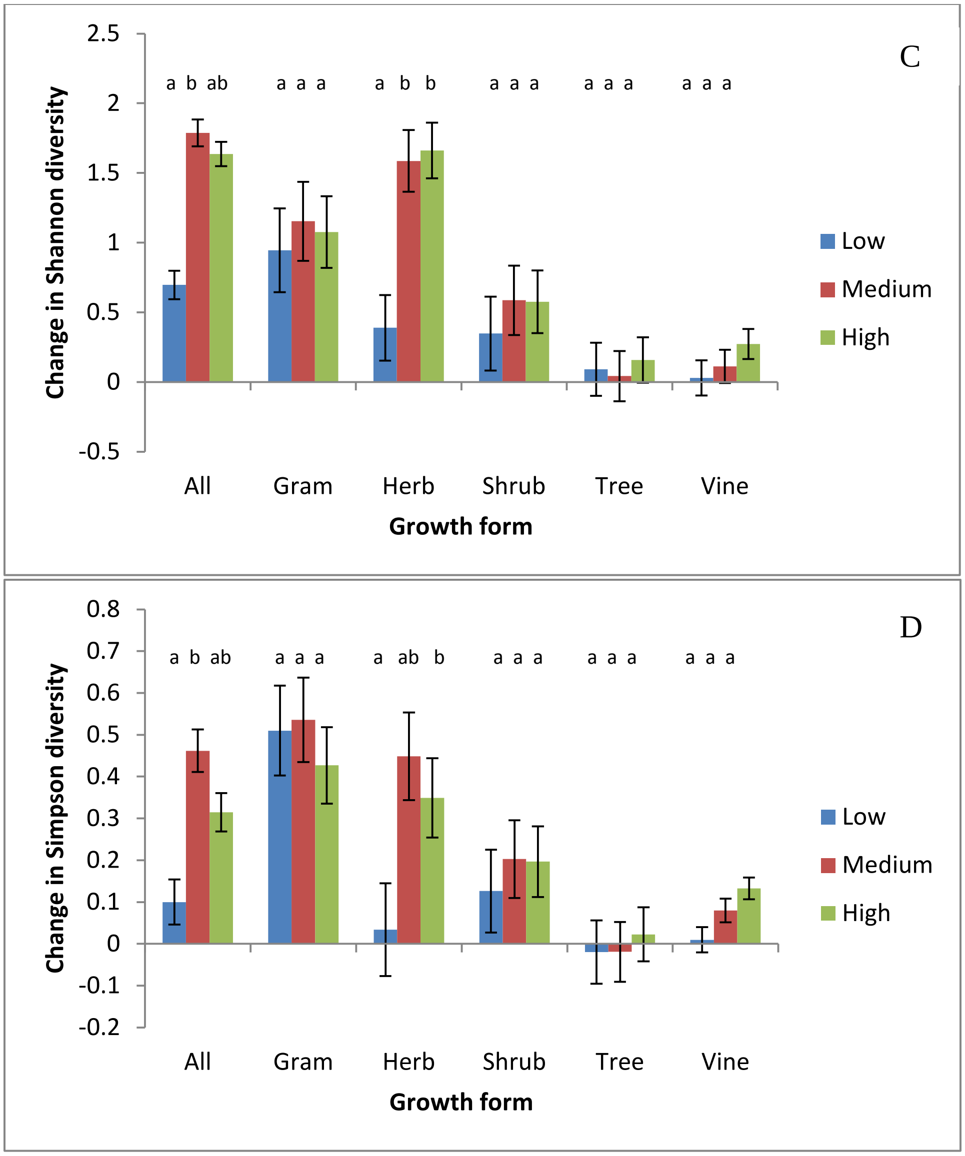
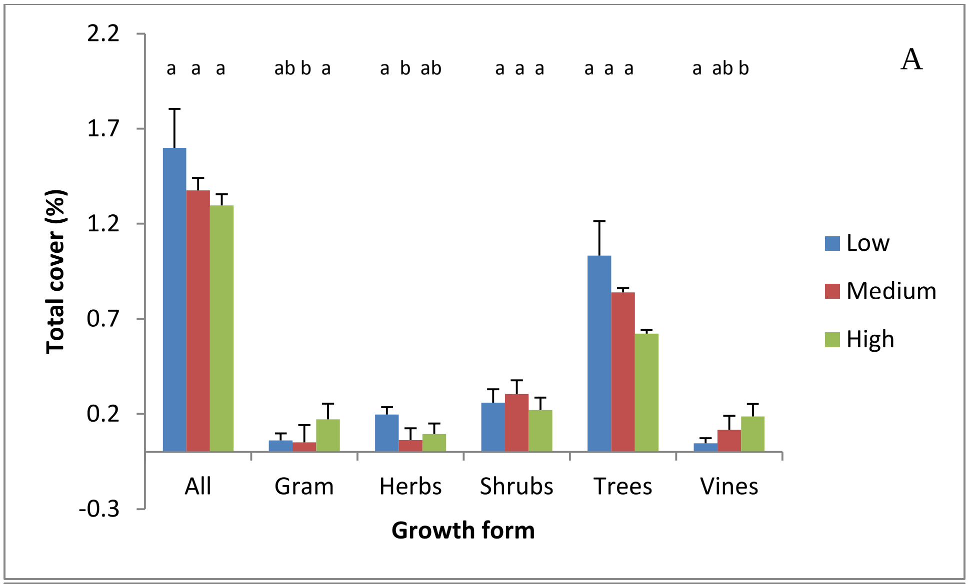
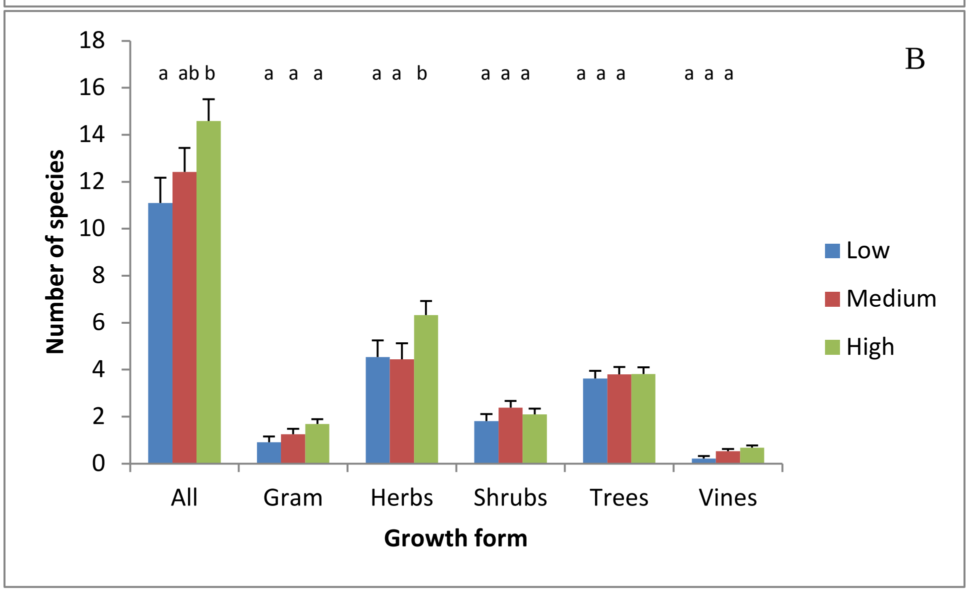


Figure 5: Percent changes in cover (A), richness (B), Shannon diversity (C) and Simpson's diversity (C) (by growth form) in plots re-burned at different degrees of severity (low, medium and high) in Linville Gorge Wilderness Area, North Carolina.


A
Total cover (%)
2.2
1.7
1.2
0.7
0.2
-0.3
a a a
ab b a
a b ab
a a a
a a a
a ab b
All
Gram
Herbs
Shrubs
Trees
Vines
Growth form
Low
Medium
High


B
Number of species
18
16
14
12
10
8
6
4
2
0
a ab b
a a a
a a b
a a a
a a a
a a a
All
Gram
Herbs
Shrubs
Trees
Vines
Growth form
Low
Medium
High

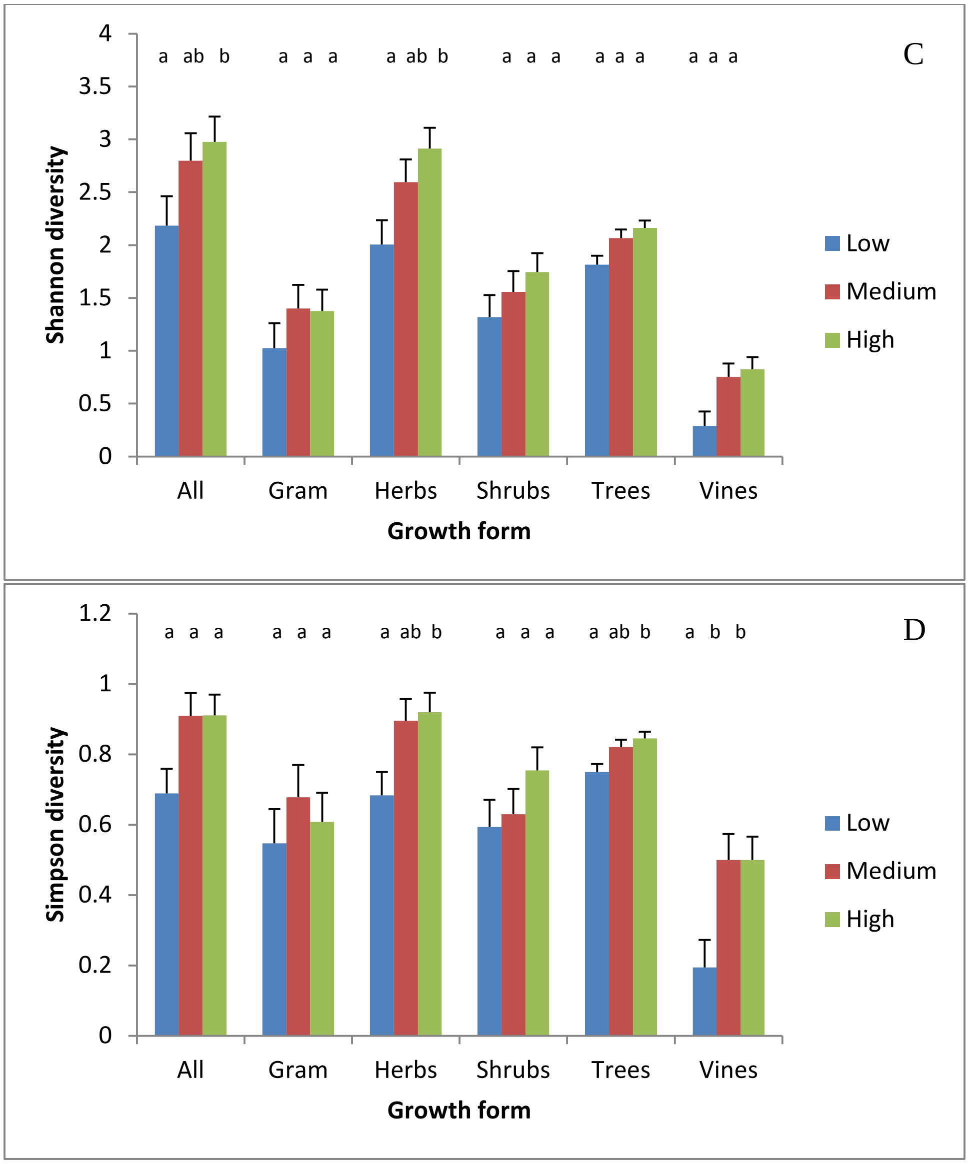


Figure 6. Current cover (A), species richness (B), Shannon Diversity (C) and Simpson diversity D) (by growth form type) in plots re-burned at different degrees of severity (low, medium and high) in Linville Gorge Wilderness Area, North Carolina.

The relationship between pre-fire shrub cover and post-fire tree cover in sites burned twice was weak. However, it appears that lower severity second fires tended to occur in sites with low pre-fire shrub cover (e.g. cover values <1). Post-fire, these plots generally had higher tree-cover than those with higher pre-fire shrub cover (e.g. cover values between 1 and 5), which tended to burn at medium or high severities (Fig. 7) There was no apparent relationship between re-burn severity on hardwood or Table Mountain pine regeneration, nor re-burn severity appear to influence exotic plant invasion.

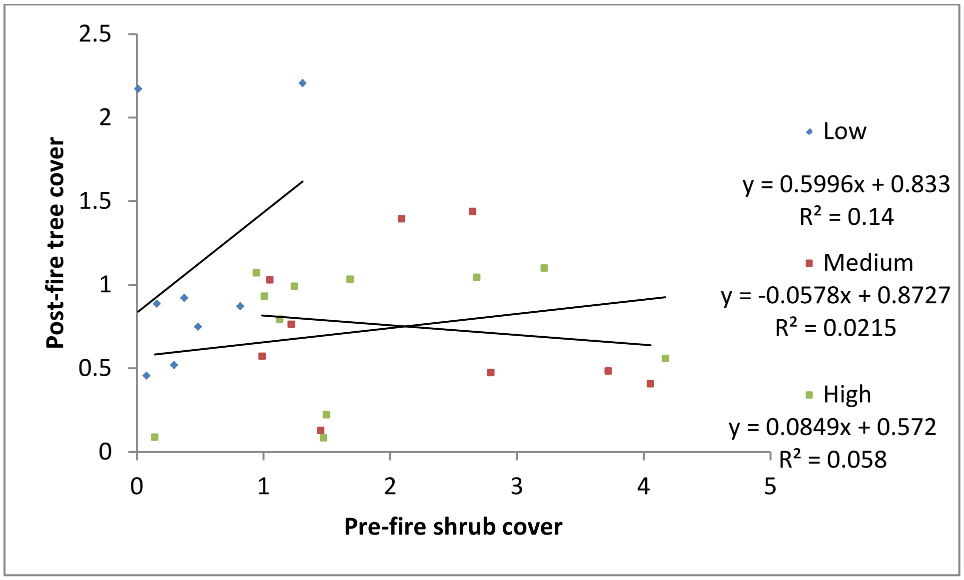


Figure 7: Regression analysis showing the relationship between pre-fire shrub cover and post fire (2x) tree cover in plots burned at different degrees of severity (low, medium and high), for shrubs (top) and trees (bottom) in Linville Gorge Wilderness Area, North Carolina.

Nonmetric Multidimensional Scaling outputs of plots burned once or twice, relative to unburned plots (Fig. 8) do not strongly indicate that fires had substantial effects on woody plant composition. Despite the significant differences in cover, richness and diversity reported above, point clouds for each of the 3 burn classes largely overlapped. Some separation, however, occurred along ordination axes due to differences in the relative covers a handful of common species. For axis 1, the most influential species were *Acer rubrum* and *Nyssa sylvatica*. For axis 2 they were *Kalmia latifolia* and *Vaccinium pallidum*, and for axis 3 they were *Acer rubrum* and *Kalmia latifolia*.

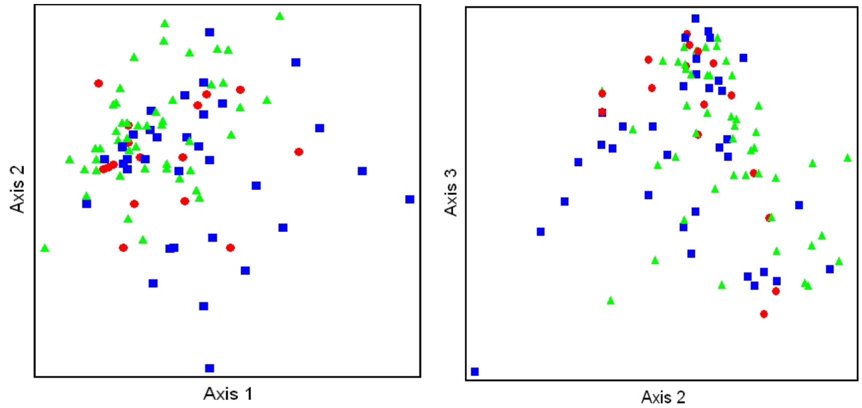


Figure 8. Nonmetric Multidimensional Scaling (NMS) biplots comparing the woody plant community assembly of unburned plots (red circles), plots burned once (green triangles), and plots burned twice (blue squares), in Linville Gorge Wilderness Area, North Carolina. All rare species removed from analysis.

Table 6. Nonmetric multidimensional scaling output showing the most influential woody plant species along each axis in the model.

Axis 1	r	Axis 2	r	Axis 3	r
Acer rubrum	-0.410	*Kalmia latifolia*	0.554	*Acer rubrum*	0.558
Nyssa sylvatica	0.421	*Vaccinium pallidum*	0.464	*Kalmia latifolia*	-0.537

Species Pearson and Kendall Correlation with Ordination Axes

Discussion

All wildfires resulted in extremely heterogeneous patterns of fire severity across the landscape ranging from low levels of litter consumption to total litter and duff consumption and complete midstory and canopy mortality. R*d*NBR values were strong predictors of fire effects on soils and vegetation. Use of R*d*NBR to model our Composite Burn Index was validated by a strong correlation between the two variables (r^2>0.76), thus allowing us to develop a map of predicted CBI values in all burned areas (fig. 2).

The distribution of CBI values over environmental gradients was similar in areas burned both once and twice. Conditions of high severity were most common on ridges and south and west facing slopes. These results suggest that landscape heterogeneity is a major driver of spatial patterns of fire severity regardless of fire history and cover type. Elevation was not strongly related to severity, although the highest elevations had the most severe fires. This trend may have occurred as fires ran upslope, became intense on ridges, and residence time was long because fires could not move upslope. Low-severity fires tended to occur on slopes that were either flat or extremely sharp. Flat slopes may have also been those that were of lowest elevation and moist, producing cooler and less severe fires. Steep slopes, on the other hand, supported fast-moving fires that moved through an area without consuming large portions of the forest floor. As should be expected, moist sites burned at lower severity than did dry sites. Although not significant, there seemed to be a threshold value at TMI=1 where severity, as expressed by CBI, was higher at lower TMI values but unchanged as moisture increased.

In study areas burned a second time, patterns of severity along these gradients were almost always higher than in areas burned once (Pinnacle 1x vs. Pinnacle 2x / Dobson Knob 1x vs. Shortoff Mountain 2x). CBI values from the Sunrise Fire were similar to those burned once in the Pinnacle Fire but generally a little lower than those burned in the Dobson Knob Fire. We propose that severity is higher in areas burned a second time because areas burned recently have greater insolation and higher temperatures, increased amounts of dead and downed woody fuels, abundant regeneration of conifers, and a dense layer of sprouting ericaceous vegetation. If the second fire is soon enough after the first, the forest floor may not have had sufficient time to reach preburn levels, thus exposing soil and creating conditions of greater severity.

Likely due to their differences in landscape position and pre-fire vegetative composition, as described above, thermic oak communities generally exhibited a greater response to burning than did acidic cove communities. These responses were most pronounced in areas subjected to two burns, as we observed significant changes in total cover, graminoid cover, shrub cover, and tree cover in these plots. They corresponded to alterations (increases) in plant species richness and diversity in these same plots. The growth form types affected most positively by repeated fire were graminoids and herbs, which is not surprising considering that these groups were largely comprised of ruderals and ephemerals which would be expected to respond quickly – if only temporarily – to such a disturbance. Fire severity correlates strongly with decreases in overstory basal area (Wimberly and Reilly 2007) due to the mortality of overstory trees. These decreases in tree cover, coupled with reductions in shrub cover, likely facilitated the establishment of these light-demanding species (Arthur et al. 1998; Kuddes-Fisher and Arthur 2002). Thermic oak plots burned once responded in a similar fashion to those burned twice, but the changes were generally of smaller magnitude (and often statistically insignificant). These findings suggest that single or once-repeated burns, regardless of severity, may not significantly alter thermic oak plant assemblages in the long term. These findings corroborate observations from other studies (Arthur et al. 1998; Elliott et al. 1999).

Response to fire was more variable in the acidic cove community, and the relationship with fire history (1x vs. 2x) was less evident. This could be due to the fact that plots in this community generally burned at lower severity than those in the thermic oak community and/or reductions in statistical power due to a smaller sample size. In general, however, the largest changes in cover, richness and diversity for burned plots in this community type occurred for ruderal graminoids and herbs – the same pattern that was observed for the thermic oak community. This again suggests that such alterations, if they did occur, were temporary.

The significant reductions in tree cover in unburned plots, in both the acidic cove and thermic oak community, reflect substantial overstory mortality that occurred due a combination of combination of drought and pine beetle infestation during the study period.

At spatial scales of $1m^2$ or smaller, the increases in species richness for plots subjected to a high severity second burn are consistent with the observed increases in richness of small-statured non-woody plants in plots described above. In some plots, this trend also reflects an increase in tree and shrub species regeneration, as mineral soil exposure and increases in solar insolation caused by high severity fires created the requisite seedbed and light conditions for early- and mid-successional trees and shrubs to germinate from the seed bank. The inherent heterogeneity of these communities, which may have contributed to their heterogeneous patterns of re-burn severity, may explain why similar trends were not observed at larger scales. Species richness (including plots that never burned) did not differ substantially at any spatial scale prior to the first burn. The divergence between re-burned and unburned plots that occurred after the second burn reflects increases in large-scale community heterogeneity caused by burning (Reilly et al. 2006a; Reilly et al. 2006b).

Re-burning facilitated the establishment of graminoids and herbs – a pattern that was particularly evident in plots re-burned at medium or high severity. This again suggests that ruderals were the species group most adept at capitalizing on the pulse of resource availability caused by the fires (Arthur et al. 1998). Increases in tree and shrub richness and diversity, despite decreases in cover for these two growth forms, suggest that burning promoted woody regeneration as well. A relationship between woody regeneration and the different levels re-burn severity, however, was not clearly evident.

The relationship between pre-fire shrub cover and post-fire tree shrub cover suggests that shrubs influence fire severity. Shrubs, especially species such as mountain laurel (*Kalmia latifolia*), are a major component of the live fuels in these community types. They often act as ladder fuels, which can increase flame heights and kill or cause damage to overstory trees (Stottlemeyer et al. 2006). This likely explains why plots with low pre-fire shrub cover burned at lower severities and had higher post-fire tree cover than those burned at medium or high severity.

While there was evidence that burning facilitated woody species, results of the multivariate analyses are less compelling. These communities were highly heterogeneous prior to burning, and they remained so after being burned 1x or 2x. There is evidence, in the form of slightly larger NMS point clouds, that burning resulted in an increase in community heterogeneity. This is not surprising, since the results for Objective 1 and Objective 2 clearly indicate the high degree of variability in fire severity across the landscape, as well as differences in community response to the number and/or severity of burns. The apparent positive relationship between number of burns the relative cover of *Nyssa sylvatica*, along with the negative relationship observed for *Kalmia latifolia*, are supported by observations from the field. *Nyssa sylvatica* was observed to sprout prolifically after fire, although it remained a relatively minor component of the community. High severity fires have been shown to kill *Kalmia latifolia* (Waldrop and Brose 1999). Aside from these changes, however, it is clear that the fires did not substantially alter plant community assembly in these sites.

Conclusions

- Five wildfires that burned in the Linville Gorge of North Carolina between 2000 and 2008 created extremely variable conditions of burn severity.

- Both R*d*NBR data from remote sensing and a composite burn index (CBI) that we derived from field data proved to be exceptionally accurate at predicting on-the-ground differences in fire severity. This allowed us to use GIS data to produce an accurate map of fire severity in three landscapes that were burned once and two landscapes that were burned twice.

- CBI followed similar patterns at all fires. The most severely-burned areas were on the higher elevations, probably ridges, on slopes of moderate steepness, and facing to the south, southwest, west, or northwest.

- The effects of 1 burn vs. 2 burns were more pronounced in the thermic oak community than in the acidic cove community.

- Shrub and tree cover decreased at all levels of re-burn severity. Herb richness and total richness were highest in the highest level of re-burn severity.

- Significant changes (increases) in species richness occurred only at scales $< 1\ m^2$, which is likely a reflection of large-scale community heterogeneity and/or heterogeneous patterns of burn severity.

- Woody plant community assembly was not substantially affected by burning, nor did burning appear to promote exotic plant invasion.

Appendix 2

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