United States Department of Agriculture

Response of Smooth Rock Skullcap (*Scutellaria saxatilis*), a Globally Rare Plant, to Fire

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Forest Service | Northern Research Station | Research Paper NRS-28 | April 2015

Abstract

Scutellaria saxatilis Riddell (smooth rock skullcap or rock skullcap, hereafter abbreviated as SRS), a herbaceous perennial in the mint family, is a globally rare (G3) plant. In West Virginia, SRS is categorized as an S2 species (imperiled and at high risk of extinction due to a very restricted range, very few [<20] documented occurrences, or steep declines). The purpose of this study was to determine the effects of fire on SRS in West Virginia. Two forested sites (70+ years of age) within the Monongahela National Forest with no evident disturbance and with SRS populations of >1,000 individuals were selected, one in a burn area and the other in a nonburn area. Sites were sampled in early September of 2008 and 2009 (pre-burn) and 2010 and 2011 (postburn). The prescribed burn occurred in April and early May of 2010. A generalized linear mixed model with repeated measures and a spatial covariance matrix was used to determine the effects of the burn on SRS cover and associated variables including total vegetation cover, species diversity, bare ground, and litter cover. Bare ground cover increased and litter cover decreased in 2010 in response to the fire. Control and pre-burn sites did not differ significantly in terms of SRS cover over the 4-year period. The cover of SRS increased significantly in 2010 (first year postburn) compared to both pre-burn years, but decreased to pre-burn levels by 2011. Total cover of other understory vegetation increased significantly in 2010 and continued at 2010 levels in 2011 at the burn site. Thus, SRS has a temporary positive response to prescribed fire, but an increase in other ground vegetation may prevent a sustained positive response.

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Cover Photo

Smooth rock skullcap, also known as rock skullcap, in flower. Photo by Ron Polgar, Monongahela National Forest.

Manuscript received for publication 10 November 2014

Published by:

U.S. FOREST SERVICE
11 CAMPUS BLVD SUITE 200
NEWTOWN SQUARE PA 19073

April 2015

For additional copies:

U.S. Forest Service
Publications Distribution
359 Main Road
Delaware, OH 43015-8640
Fax: (740)368-0152
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INTRODUCTION

Smooth rock skullcap (*Scutellaria saxatilis* Riddell; Fig. 1), also known as rock skullcap and hereafter abbreviated as SRS, is a herbaceous perennial in the mint family (Lamiaceae), with decumbent stems which differentiate it from most other more common skullcaps. Its violet or white flowers are in one-sided terminal racemes, and the stem is spreading glandular in the lower part, mostly glabrous in the middle, and spreading hairy and glandular in the inflorescence (Epling 1939, Gleason and Cronquist 1993). Some populations in West Virginia may tend to be more glabrous and eglandular than in other states (Epling 1939, Strausbaugh and Core 1977). SRS produces flowers as early as July and as late as September. Reproduction is via seeds or vegetatively via threadlike stolons (Epling 1939, Gleason and Cronquist 1993). SRS is part of the *Scutellaria ovata* species-group, an American group in which the nutlets lack hairs and the corolla is relatively blue in color; slender stolons are often present at the base of the plant (Paton 1990). Members of this group are distributed in the eastern and south-central United States and Mexico and also include *S. arguta* Buckley, *S. cardiophylla* Engelm. & A. Gray, *S. pallidiflora* Epling, and *S. ovata* Hill. SRS is also genetically and morphologically similar to the *S. havanensis* species-group, which comprises spreading or weakly ascending herbs that have blue or purple corollas and are distributed in Florida, the West Indies, and Mexico (Paton 1990).

SRS may show a preference for rich, moist, and rocky woods (Olson et al. 2004, Terrell 1970) with dense shading, and is thought to have a negative response to a loss of forest canopy (Dolan 2004, Olson et al. 2004). However, whether adequate soil moisture, shade, or some other factor is more important is not clear. Some populations are located in more open areas, such as along roads (Wofford and Dennis 1976), suggesting that a dense canopy is not a requirement for successful establishment. The requirement for high moisture also may be inaccurate; populations of SRS are associated with dry, limestone bluffs, crevices, and exposures in Giles County, VA (Cooperrider and Thorne 1964), dry hills in Ohio, and arid cliffs in Kentucky (Epling 1939). Yet SRS is also found in moist hemlock-beech-

Figure 1.—*Scutellaria saxatilis* in flower. Photo by Ron Polgar, Monongahela National Forest.

rhododendron forests in Giles County (Cooperrider and Thorne 1964). In addition, SRS is found in damp to dry soil of deciduous rocky woods along Skyline Drive in Warren County, VA (Mazzeo 1972). SRS has also been associated with the rare upland red spruce community type in West Virginia (Byers et al. 2010). SRS does not appear to compete well with other plant species (Dolan 2004), and rockiness appears to be the most shared habitat characteristic described in the literature.

SRS is a globally rare (G3) plant species. A G3 ranking is defined as vulnerable and at moderate risk of extinction due to a restricted range. Species with this ranking have relatively few known populations (usually <80) and show evidence of recent and widespread decline. SRS is currently found in 16 states in the eastern United States. It is listed as reported without persuasive evidence of its occurrence in Arkansas, District of Columbia, New Jersey, and South Carolina and as possibly extirpated in Delaware. SRS is listed as state endangered in Alabama, Georgia, Indiana, Maryland, North Carolina, and Pennsylvania. It is listed as threatened in Ohio, Tennessee, and Virginia, and as rare in Kentucky and West Virginia (NatureServe 2014b). This species was also noted in Missouri in Miller County in 1883 by an amateur botanist, but the species identification has been questioned (Bush 1926) or the species may now be extirpated. The Hoosier (Indiana), Monongahela (West Virginia), and Wayne (Ohio) Region 9 National Forests list SRS as a Regional Forester's Sensitive species. In 1966, SRS was described as frequent on wet banks throughout West Virginia (Clarkson 1966). In 2004, SRS was categorized as an S1 species (≤5 known occurrences in the state or only a few remaining individuals) in West Virginia (Dolan 2004). In 2005, however, SRS was ranked as S2 (E. Byers, West Virginia Division of Natural Resources, pers. comm.), which is defined as imperiled and at high risk of extinction in West Virginia due to a very restricted range, very few (<20) documented occurrences, or steep declines (NatureServe 2014b). It is likely that this new ranking reflects the increase in inventory effort for this species in West Virginia (E. Byers, pers. comm.).

Prescribed burns by Native Americans and early European settlers were used to maintain open areas, savannas, and oak-dominated forests (Abrams 1991, Brose et al. 2001). Starting in the 1920s, a policy of fire suppression was widely adopted and has been blamed for the mesophication of Eastern forests, a positive feedback cycle in which shade-tolerant and fire-sensitive species increase in abundance relative to shade-intolerant species (Brose et al. 2001, Nowacki and Abrams 2008). Prescribed burns are now being used more frequently in Eastern ecosystems to restore oak savannas and woodlands as well as improve oak regeneration in forested areas. The latter use of fire is often combined with shelterwood harvesting (Barnes and Van Lear 1998, Brose and Van Lear 1998, Brose et al. 2001). Two species dependent on shale-barren communities, which may be dependent on fire to prevent woody species encroachment, are *Trifolium virginicum* Small ex Small & Vail (Kates Mountain clover), another Regional Forester's Sensitive plant species in West Virginia, and *Boechera serotina* (E.S. Steele) Windham & Al-Shehbaz (shale-barren rockcress), a federally endangered species in West Virginia. Whether fire is needed to maintain shale-barren communities located in harsh environmental conditions is uncertain. The loss of species in these communities is generally associated with the loss of their habitat to development (e.g., roads, housing, urban sprawl), not necessarily to a lack of fire (Bartgis 1987, Nott 2006).

In contrast, *S. floridana* Champm. (Florida skullcap) is an example of a federally threatened rock skullcap found in longleaf pine-grassland habitats. Fire suppression and silvicultural activity are thought to be reasons for its decline (Van Lear et al. 2005, Walker 1993). Whether or not SRS' habitat loss is more closely associated with a lack of fire (like *S. floridana*) or development (possibly similar to *T. virginicum* and *B. serotina*) is not known. SRS' apparent preferred habitat of rocky wet slopes may not be fire dependent, though oaks do dominate the sites and mesophication of these oak forests may be connected to SRS' decline. However, there is also weak evidence that SRS may decrease in abundance in response to a canopy opening (Dolan 2004, Olson et al. 2004). Given that fire may increase openings in the subcanopy and canopy layers of forests (Canham et al. 1990), the impact of fire on SRS may be negative, depending on the degree of canopy cover reduction.

The purpose of this study was to determine the effects of fire on SRS in West Virginia. Fire typically reduces the cover of dominant species (Hannah 1987) and may temporarily increase resources (nutrients; Certini 2005). However, fire may also reduce canopy cover by causing subcanopy or canopy tree deaths, thereby increasing the light reaching the forest floor through canopy gaps (Canham et al. 1990). Because there has been some evidence of a negative response of SRS to a decrease in canopy cover (Dolan 2004, Olson 2004), we tentatively predicted that SRS cover would decrease in response to fire.

METHODS

Study Area

The study was located in Pocahontas County in the Greenbrier Ranger District of the Monongahela National Forest (Fig. 2). The site is part of an oak forest regeneration project that is associated with a nearby savanna restoration project called the Ramshorn Prescribed Burn project. There, SRS is found in about 45 separate locations (patches of one to thousands of individuals ≥5 m apart) in planned prescribed burn areas, as well as in 80 separate locations in adjacent nonburn (control) areas. The savannas planned for restoration are located along the ridges, but both ridges and slopes were intended to be burned, hence the concern over the possible impact on the SRS populations. Site selection was made from available areas with >1,000 individuals, and, thus, our statistical inference level is for sites with more than 1,000 SRS individuals. There were two to select from within the locations scheduled to be burned during the time of the study, one that was south- to southwest-facing and the other east- to northeast-facing. We found two sites in the nonburn areas that were close in age (≥70 years of age) to the proposed burn sites and that faced the same directions. The east- to northeast-facing slope of the nonburn area was located in a younger forest than the corresponding paired burn site, and had a very narrow area of SRS, so this site did not qualify as a replicate site. Thus, only two sites composed of forests ≥70 years of age, one in the burn area and the other in the nonburn area, were selected. Both had south- to southwest-facing slopes. Neither showed evidence of disturbance. The burn site was located along a ravine off Stony Run in the Chestnut Ridge area; the control site was located in a ravine off Sutton Run, also in the Chestnut Ridge area, but about 5 km from the burn site. We thus consider these sites separate SRS populations or element occurrences as defined by NatureServe (2014a).

The sites were sampled the first week of September in 2008, 2009, 2010, and 2011, and the prescribed burn occurred in April and early May of 2010. Fire intensity on the slopes was not monitored. Mortality of some large canopy trees on the slope and on the ridge indicates that the fire was a relatively hot burn in patches.

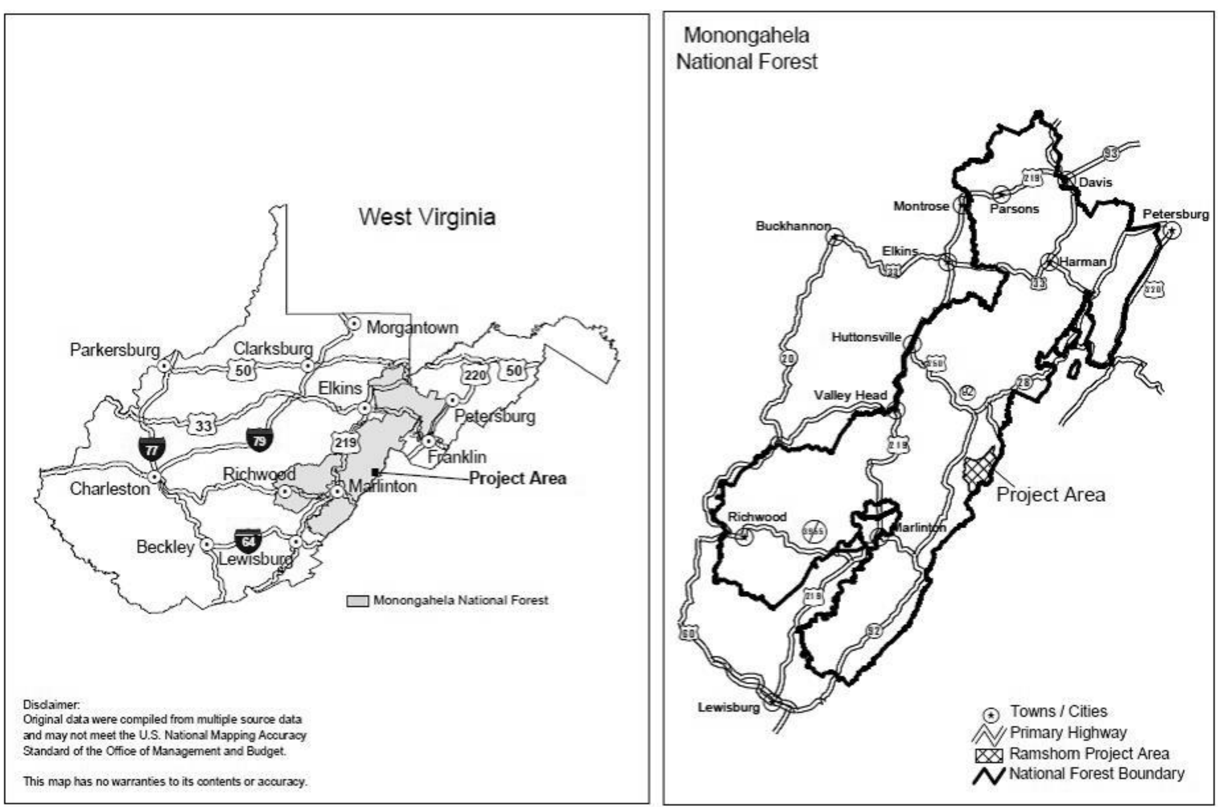

Figure 2.—Study area showing general location within the state and general location within the Monongahela National Forest.

Sampling Design

A stratified-random sampling design with two plot sizes was used at both sites (Fig. 3). A 100-m transect was set up in a central location along the contour of the slope where SRS was relatively abundant. Universal Transverse Mercator (North American Datum 83) coordinates were taken at the beginning and end of the transect, and a transect bearing was noted. It is Monongahela National Forest's policy not to publish location information of threatened species, but we may be able to provide this information upon reviewed request. Slope inclination was 20-25 percent for both sites. Every 10 m along each transect starting from the beginning, a 1-m^2 circular plot area was marked with a stake chaser and sampled systematically. At random distances >3 m and <50 m, one circular quadrat was placed uphill perpendicular to the transect and one circular plot was set up downhill perpendicular to the transect. One of the uphill or downhill plots was 1 m^2 in area and the other one was 10 m^2 in area. The larger and smaller plots alternated in terms of being upslope or downslope, with the first selection of upslope or downslope being randomly chosen. Thirty plots were sampled at each site in each year except 2011. In 2011, three plots were not sampled at the burn site to avoid unsafe conditions due to their proximity to dead and falling canopy trees. This sampling method was selected because it is among the best at documenting rare species as well as the dominant plant species (Huebner 2007).

Cover was estimated in each plot for all herbaceous plants rooted in the plot and all foliage of vines and shrubs falling within the plot boundaries with no requirement that they be rooted in the plot. Cover values of species in the 10-m^2 plots were converted to cover per 1 m^2 for the calculation of relative cover and importance values for each species and for subsequent statistical analyses. Vegetation cover data for each plot could potentially sum to >100 percent. Percent cover of litter, percent cover of bare ground, and litter depth were also determined at each plot. These data were collected in conjunction with rock, moss, and coarse woody debris cover, which summed to 100 percent. From these data, SRS cover, total vegetation cover excluding SRS, and diversity using the Shannon (H´) and Simpson indices

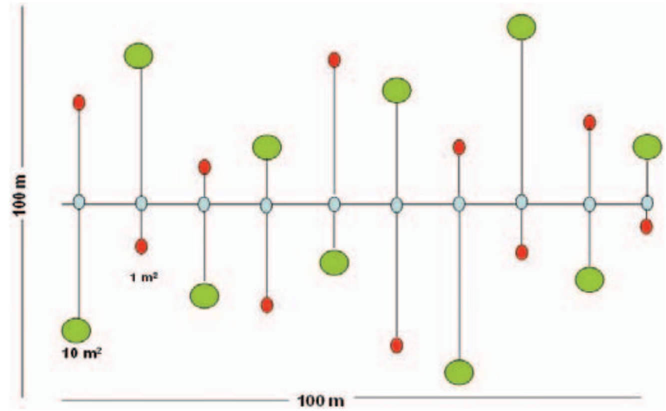

Figure 3.—Stratified-random sampling design using a 100-m-long transect.

excluding SRS were determined. Most SRS plants were still in flower and some were in fruit each year at the time of sampling. Because of the vegetative growth associated with SRS, cover estimates were considered more accurate than stem counts. Nonetheless, where cover was high, stem counts also tended to be high, based on an initial comparison of cover and stem counts measured in 2008 at both sites. Nomenclature of plant species follows the Interagency Taxonomic Information System (2014).

Statistical Analyses

Because of the lack of site replication and the focus on population-level effects rather than community-level effects, the plots were used as replicates to determine the effect of the burn on SRS. Such an approach, though not ideal (Hurlbert 1984, Palmer 1987), is not uncommon in population studies of threatened species or in large-scale community studies (Elzinga et al. 2001, Keith 2000, Oksanen 2001). Nevertheless, we addressed the potential for spatial autocorrelation due to pseudoreplication by including an autoregressive (AR(1)) covariance matrix associated with plots as a random variable. A generalized linear mixed model, which included both the spatial covariance matrix and an AR(1) covariance matrix (for the year-to-year or repeated sampling), with a lognormal distribution and identity link function (GLIMMIX, SAS 9.4; SAS Institute Inc., Cary, NC) was used to determine the effects of the burn on SRS, total vegetation, bare ground, and litter cover. A Gaussian distribution with an identity link function was the best fit for comparing the effect of the burn on herb, shrub, and vine diversity. The same models were used to compare each of the variables (herb, shrub, and vine diversity; and SRS, total vegetation bare ground, and litter cover) at the control site across the 4 years. Plot was the only random effect and the fixed effects were burn or no-burn and year. The selection of AR(1) as the covariance structure and the distributions were determined from best fit statistics, including deviance, log likelihood, and Pearson's chi square, and by evaluating each variable under different distributions with univariate statistics. Pearson residuals were evaluated as well as residual vs. predicted value plots. All means were compared by using a Tukey's multiple comparisons test. We conducted the same analyses for all plots and separately for just the plots containing SRS.

Using the above analyses we chose not to do a direct statistical comparison of burn-site plots with control-site plots but instead to use the control site as a separate measure of other possible factors not associated with fire that could be affecting SRS over the study period. We then analyzed these data by using a Before-After-Control-Impact (BACI) analysis for each variable to determine if there were potential significant site (control vs. burn) conditions over the period of the study unrelated to the burn treatment that may confound our conclusions about the effect of fire. We used the same model statements within our GLIMMIX models above, including the covariance matrix for year-to-year sampling of the same sites and the spatial covariance matrix for possible pseudoreplication. But we also included the control site and the site type (burn or control) and year (2008, 2009, 2010, or 2011) interaction term and used contrast statements to compare site type and year combinations. We first compared the pretreatment years (2008, 2009), using all data from both sites. Second, we analyzed data from the nonburn years: all data from the pretreatment years as well as data from 2010 and 2011 from the control site. We then compared the postburn years, which included both 2010 and 2011 data from the burn site, with the nonburn-year data (2008 and 2009 from the burn and control site, and 2010 and 2011 only from the control site). Finally, we contrasted only data from 2010 from the burn site with all nonburn-year data. Significant differences among pretreatment years or nonburn years would indicate that there may be important site or temporal variables unrelated to the burn. This analysis assumes that site type and year variation are equal across sites and years (Schwarz 2002).

RESULTS AND DISCUSSION

Bare Ground and Litter Cover

The prescribed burn affected the burn site by increasing bare ground and decreasing litter. At the burn site, bare ground cover differed significantly ($F = 7.39$, $P = 0.0002$) among the years. There was much more bare ground on average in 2010 than in either pretreatment year or in the postburn year 2011 (Fig. 4a). Bare ground cover at the control site did not differ significantly by year ($F = 0.12$, $P = 0.95$; Fig. 4c). Litter cover differed significantly ($F = 5.89$, $P = 0.0011$) between 2009 and both postburn years at the burn site (Fig. 4b). There was no significant difference in litter cover ($F = 0.29$, $P = 0.83$) at the control site (Fig. 4d). These trends for bare ground and litter were the same when evaluating only the plots containing SRS (Fig. 4a-d). Though not presented, rock cover increased substantially in 2011 at the burn site, which may explain the lower bare ground cover for that year. Although we defined bare ground as mineral soil and not duff, some organic material may have been included in our cover estimates of bare ground. The organic material could have burned, possibly revealing a rock layer below. It is also possible that precipitation or other small-scale disturbances may have removed a superficial layer of soil between 2010 and 2011, revealing more underlying rock. In contrast, bare ground cover was not lower in 2011 compared to 2010 at the control site.

SRS Cover

The burn site had more plots with SRS (13-15 plots) than the control site (9-11 plots). At both sites, plots that did not contain SRS in a given year generally did not gain SRS the following year; 74-95 percent of all plots with no SRS remained without SRS throughout the study period. Thus, most changes in SRS abundance at the sites are associated with the individual plots containing SRS, which supports the need to evaluate the plots containing SRS separately as well as with the other plots. A large increase in SRS abundance on the burn site

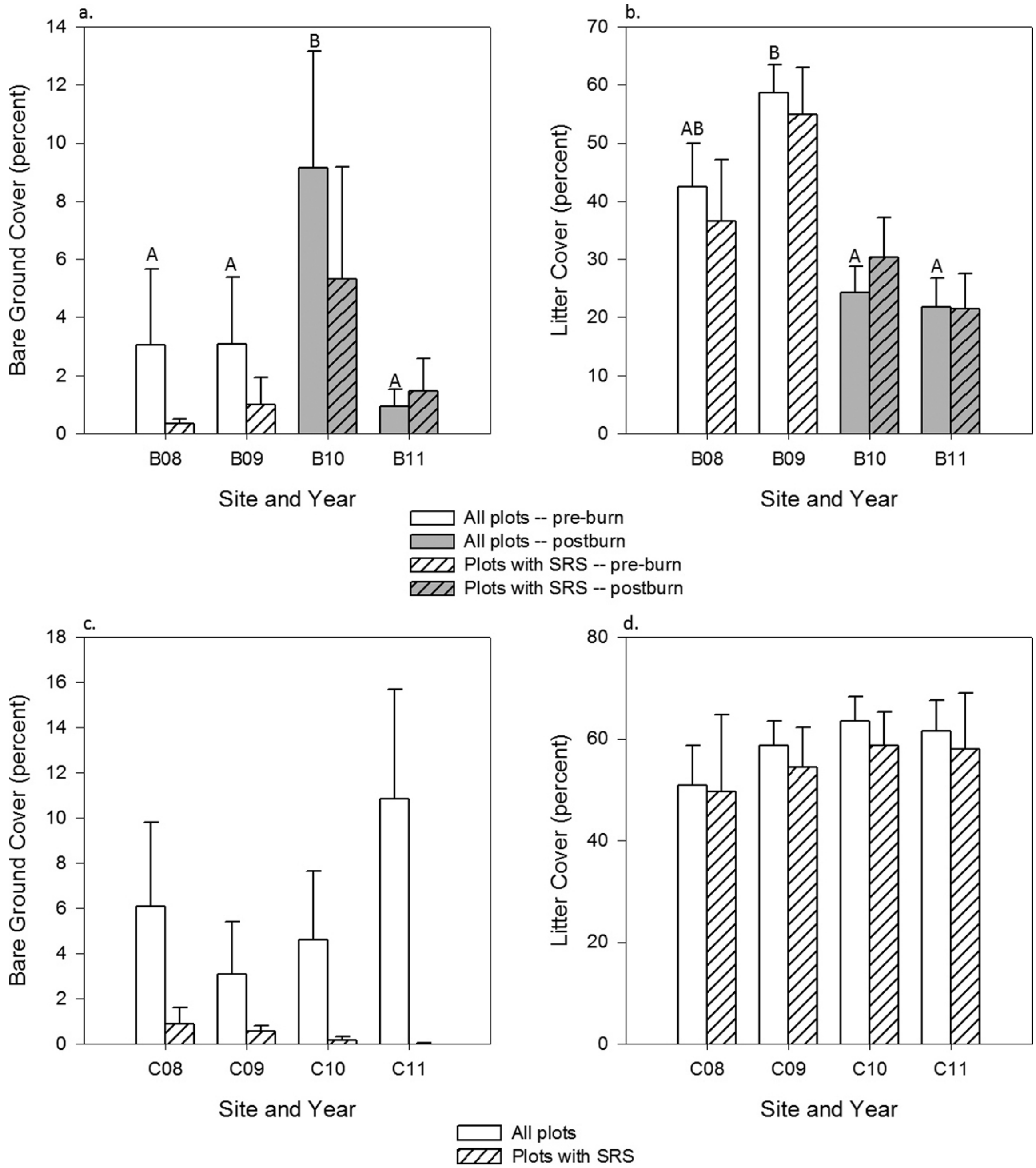

Figure 4.—Comparison, with standard errors, of (a) bare ground cover and (b) litter cover at the burn site and (c) bare ground cover and (d) litter cover at the control site. Statistical comparisons across years were made by using a generalized linear mixed model (GLIMMIX) with autoregressive covariance matrix, gamma distribution, and a log link function. Bars with different letters within the same graph are significantly different at $P < 0.05$.

occurred the first year after the burn. Abundance of SRS also increased at the control site between 2008 and 2009. There was a relatively large decrease in abundance of SRS between 2008 and 2009 at the burn site and between 2010 and 2011 at the control site (Table 1). The increase in abundance of SRS in 2009 at the control site and the decrease in abundance at the burn site in 2009 cannot be attributed to any known events, suggesting year-to-year abundance of SRS typically may be variable. The decrease in abundance of SRS at the control site in 2011 may be linked to a possible high precipitation event. According to weather data from a nearby weather station in Frost, WV, the total annual precipitation was 107.98, 117.17, 117.50, and 132.21 cm in 2008, 2009, 2010, and 2011, respectively (National Oceanic and Atmospheric Administration [NOAA], National Climatic Data Center

Table 1.—Number of SRS plots showing change in SRS abundance, by site and year; and percentage of non-SRS plots that contained no SRS the following year

Site	Total number of plots containing SRS, 2008	Increase	Decrease	No change	Proportion of plots with 0 SRS in both years	Total number of plots containing SRS, 2009
		2008-2009				
Burn	15	3	11	1	93%	15
Control	9	7	2	0	86%	11
		2009-2010				
		Increase	Decrease	No change	Proportion of plots with 0 SRS in both years	Total number of plots containing SRS, 2010
Burn		9	3	2	94%	13
Control		3	7	1	74%	10
		2010-2011				
		Increase	Decrease	No change	Proportion of plots with 0 SRS in both years	Total number of plots containing SRS, 2011
Burn		1	11	0	87%	14
Control		2	8	0	95%	9

[NCDC] 2014). Unusually high precipitation occurred in March and April of 2011 compared to the same period in previous years and could have resulted in some flooding in the ravines near the slopes of our study sites, potentially impeding the establishment of many plants on the slopes early in the growing season.

Abundance of SRS did not differ significantly among all 4 years of sampling at the burn (F = 0.31, P = 0.82) and control (F = 0.04, P = 0.99) sites (Fig. 5a and c) when evaluating all plots together. Because there was an obvious decline in SRS cover from 2010 to 2011 at both the burn and control sites (Fig. 5a and c), we also ran the models without 2011 data. After removal of the 2011 data at the burn sites, SRS cover when evaluating all burn site plots was insignificant (F = 0.42, P = 0.66), though there was a trend for 2010 (first postburn year) to have more SRS cover than 2009 (Fig. 5b). There was no similar trend for the control site (F = 0.64, P = 0.53). Evaluating only the burn site plots with SRS, and excluding 2011, did show a significant positive effect of the burn (F = 5.51, P = 0.011), with significantly higher SRS abundance in 2010 than 2009 in plots with SRS (Fig. 5b). Including only the years 2008, 2009, and 2010 in the models of only the SRS plots at the control site showed no significant difference in the years (F = 1.59, P = 0.24; Fig. 5d). The burn site had significantly greater (about twice as much) SRS cover than the control site in 2010 and 2011 (Fig. 5a-d). There was also a decline in SRS cover in 2011, though insignificant, at the control site (Fig. 5c), which may indicate that conditions associated with 2011, or just with the control site, were not optimal for SRS growth, perhaps due to a high precipitation event as discussed above. If we assume that the unknown conditions were having a negative impact on SRS at both sites, the burn appears to have continued to benefit SRS in 2011 as well as 2010 in spite of this unknown condition. We reached this conclusion because of the higher SRS percent cover at the burn site in 2011 compared to the control site. Nonetheless, the control site's 2009 SRS cover was nearly as high as the burn site's 2010 SRS cover. It is possible that we are simply looking at extreme variations in year-to-year SRS cover at two sites.

Our focus on abundance in this study, unfortunately, does not tell us much about SRS's fitness in response to the burn. It would be useful to know if the weak

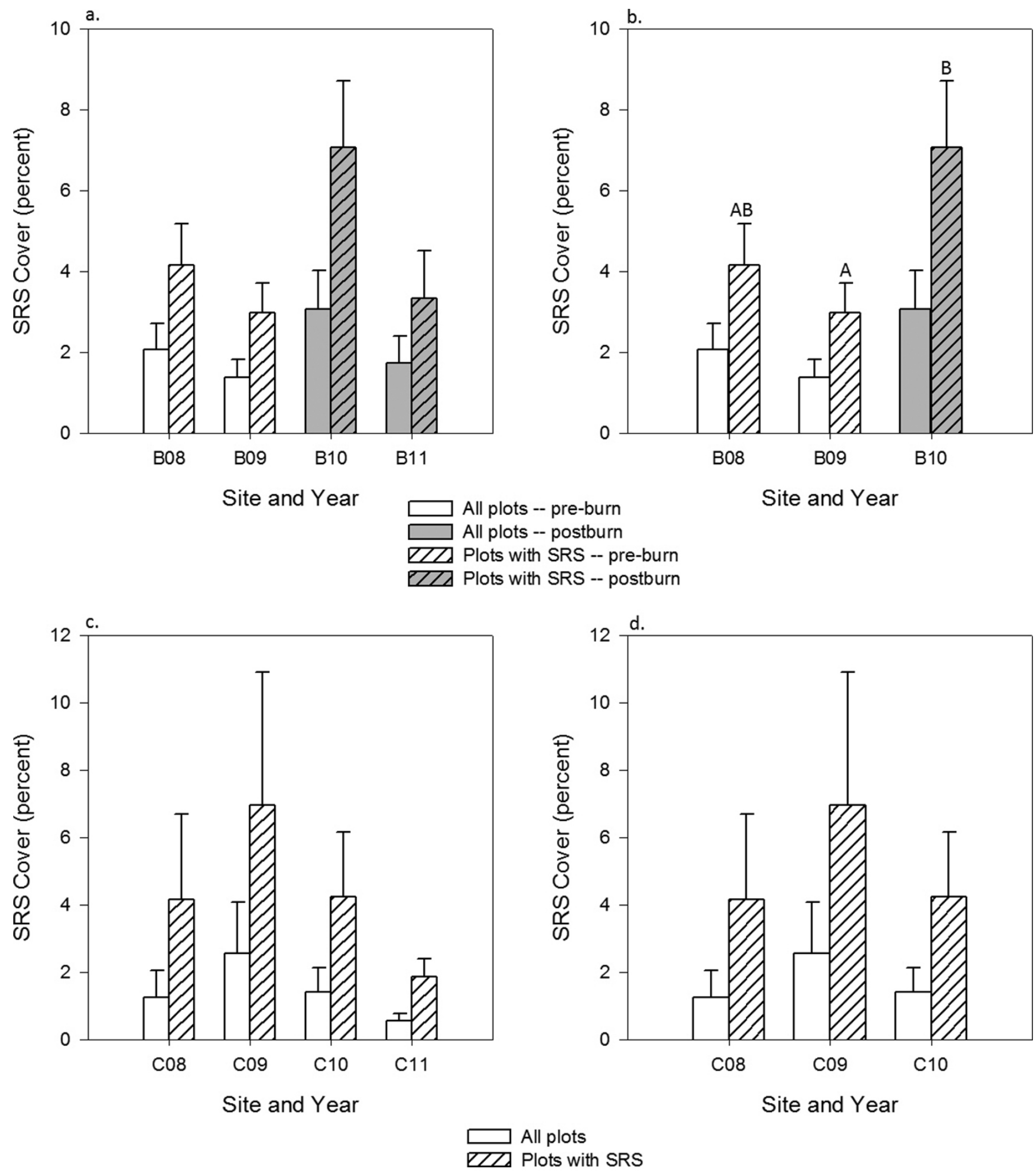

Figure 5.—Comparison, with standard errors, of SRS abundance at (a) the burn site for all 4 years, (b) the burn site removing 2011 data from the analyses, (c) the control site for all 4 years, and (d) the control site removing 2011 from the analyses. Statistical comparisons across years were made by using a generalized linear mixed model (GLIMMIX) with autoregressive covariance matrix, gamma distribution, and a log link function. Bars with different letters within the same graph are significantly different at < 0.05.

positive initial response of SRS to fire was primarily due to sprouting or if it may have also stimulated more seed production and seedling survival or even seed germination from any existing seed bank. The fact that most increases appear to be associated with the same plots suggests that most increases are due to sprouting and not new seedlings. Nonetheless, if flowering and seed set increased in response to the fire, the consequent increase in genetic diversity may be an additional reason to burn where SRS populations are located. Improved genetic integrity can make catastrophic population declines due to a pathogen or disease less likely for a rare species, assuming the species is a successful outcrosser. Other *Scutellaria* species have known pollinator

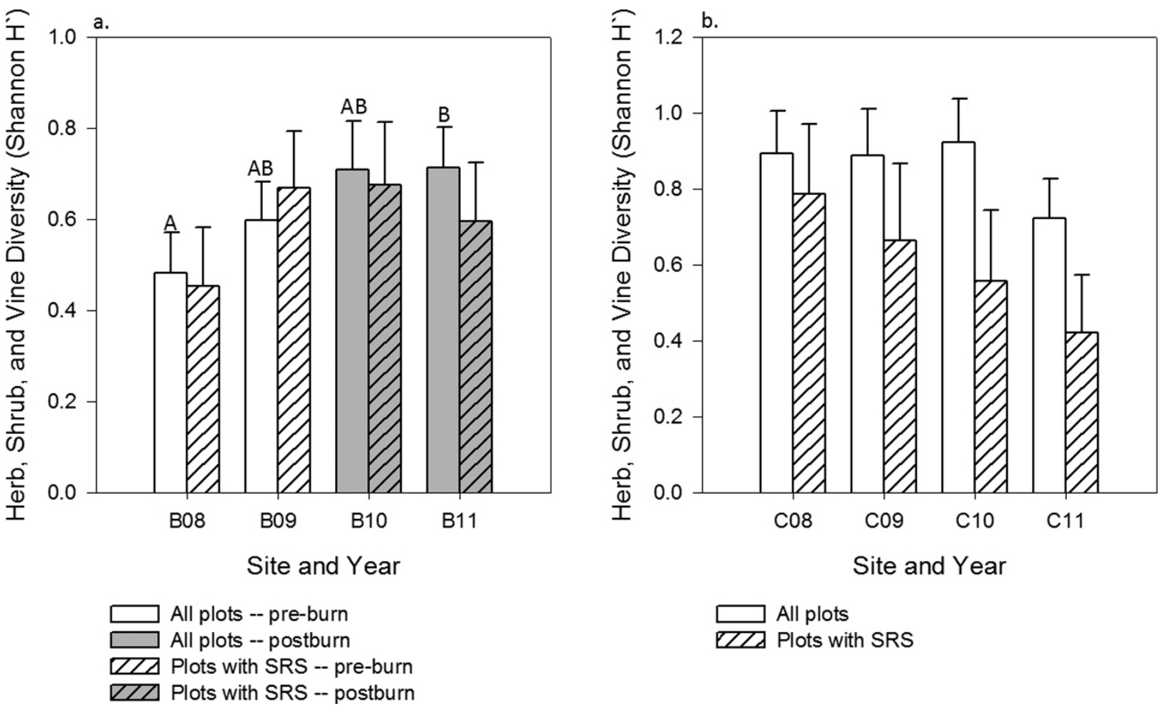

Figure 6.—Comparison, with standard errors, of species diversity (Shannon H' index) for all 4 years at (a) the burn site and (b) the control site. Statistical comparisons across years were made by using a generalized linear mixed model (GLIMMIX) with autoregressive covariance matrix, Gaussian distribution, and an identity link function. Bars with different letters within the same graph are significantly different at $P < 0.05$.

generalists and specialists, indicating that they generally outcross but also that those in specialized habitats tend to have specialist pollinators (Miller-Struttmann 2013, Pitts-Singer et al. 2002). If such specialist pollinators are rare, rare plants may further decline. There is at least one *Scutellaria* species with individuals that have only cleistogamous flowers or are a mixture of cleistogamous and chasmogamous flowers (Sun 1999). We did not see evidence of cleistogamy in SRS and did not find any mention of it in the literature; SRS's mating system needs further study. Fire may also influence flowering phenology of SRS or other species at the site. Platt et al. (1988) found that a prescribed burn in Florida occurring in April-August (i.e., growing season) delayed, but increased and synchronized, flowering of dominant species.

Species Composition

There were 26 total herb, shrub, and vine species in both 2008 and 2009 at the burn site. After the burn there were 34 species in 2010, but once again 26 species in 2011. Species diversity for herbs, shrubs, and vines collectively differed significantly ($F = 2.81$, $P = 0.044$) at the burn site among the 4 years with 2008 being less diverse than 2011, and there was a trend for increasing diversity in both postburn years compared with the pretreatment years (Fig. 6a). This significant difference in diversity is lost when evaluating only the SRS plots ($F = 0.16$, $P = 0.92$; Fig. 6a). The burn site in 2008 and 2009 shared similar species with the highest importance values and SRS remained in the top three most important species in every year. In 2010 and 2011, after the burn, weedier species, such as *Erechtites hieraciifolius* (L.) Raf. ex DC., *Rubus* L. sp., and *Phytolacca americana* L., were ranked higher in importance than in previous years, making the list of the top 10 most important species after the burn (Table 2). These species were most dominant in 2011. SRS had a lower importance ranking in 2011 compared to 2010, possibly in response to the increase in dominant weedier species. In both postburn years, but more so in 2011, there were weedy species present after the burn that were not present before the burn, including *Taraxacum officinale* F.H. Wigg., *Packera aurea* (L.) Á.Löve & D. Löve, and *Solanum dulcamara* L.

Table 2.—Top 10 species at each site during each year by importance value

Burn site					
	Pretreatment years			Postburn years	
Species	2008[†]	2009[‡]	Species	2010[†]	2011[‡]
Scutellaria saxatillis Riddle[§]	9.60*	6.98*	*Ageratina altissima* var. *altissima*	10.47*	14.27*
Aristolochia macrophylla Lam.	7.00*	7.70*	*Scutellaria saxatilis*	10.04*	3.76*
Ageratina altissima var. *altissima* (L.) King & H. Rob	1.95*	3.28*	*Erechtites hieraciifolia* (L.) Raf. ex DC	2.71*	4.49*
Arisaema triphyllum (L.) Schott.	1.77*	5.63*	*Galium triflorum* Michx.	1.95*	0.64*
Solidago curtisii Torr. & A. Gray	1.59*	1.32*	*Rubus* L. sp.	1.33*	1.52*
Galium triflorum	1.36*	2.12*	*Carex appalachica* J.M. Webber & P.W. Ball	0.67*	0.016
Eurybia divaricata (L.) G.L. Nesom	0.71*	0.70*	*Ribes* L. sp.	0.57*	0.10
Carex L. sp.	0.50*	0.13	*Solidago curtisii*	0.44*	0.62*
Impatiens L. sp.	0.31*	0.23*	*Sambucus* L. sp.	0.25*	0.10
Hamamelis virginiana L.	0.25*	0.074	*Impatiens* sp.	0.24*	0.33*
Elymus hystrix L.	0.010	0.86*	*Phytolacca americana*	0.22	0.24*
Panax quinquefolius L.	0.0	0.18*	*Pilea pumila* (L.) A. Gray	0.042	0.18*
			Rubus odoratus L.	0.064	0.16*

Control site					
	Pretreatment years			Postburn years	
Species	2008	2009	Species	2010	2011
Solidago curtisii	5.60*	3.77*	*Scutellaria saxatilis*	5.70*	2.45*
Ageratina altissima var. *altissima*	3.84*	2.77*	*Ageratina altissima* var. *altissima*	5.19*	2.37*
Aristolochia macrophylla	3.74*	3.07*	*Solidago curtissii*	4.93*	7.08*
Scutellaria saxatilis	3.71*	6.38*	*Aristolochia macrophylla*	3.60*	6.37*
Galium triflorum	2.22*	4.30*	*Impatiens* sp.	2.02*	1.76*
Laportea canadensis (L.) Wedd.	1.16*	0.0	*Galium triflorum*	1.72*	0.61*
Pilea pumila	0.68*	0.0	*Eurybia divaricata*	1.03*	0.96*
Arabis laevigata (Muhl.ex Willd.) Poir.	0.57*	0.15	*Rubus* sp.	0.59*	0.075
Eurybia divaricata	0.56*	0.67*	*Thalictrum* L. sp.	0.51*	0.86*
Ribes sp.	0.55*	0.08	*Pilea pumila*	0.41*	0.48
Impatiens sp.	0.32	3.00*	*Arabis laevigata*	0.097	0.69*
Arisaema triphyllum	0.13	1.72*	*Hydrophyllum virginianum* L.	0.0	0.57*
Hydrophyllum virginianum	0.45	0.65*			
Osmorhiza claytonii (Michx.) C.B. Clarke	0.034	0.56*			

[†] Species are listed from highest to lowest importance value at each site for the first year of the two pretreatment years and the first year of the two postburn years.

[‡] Even if a species was not among the same top 10 in the preceding year, the importance value is given. The actual top 10 for each year have an asterisk (*) by their importance values.

[§] Nomenclature follows the Interagency Taxonomic Information System (2014).

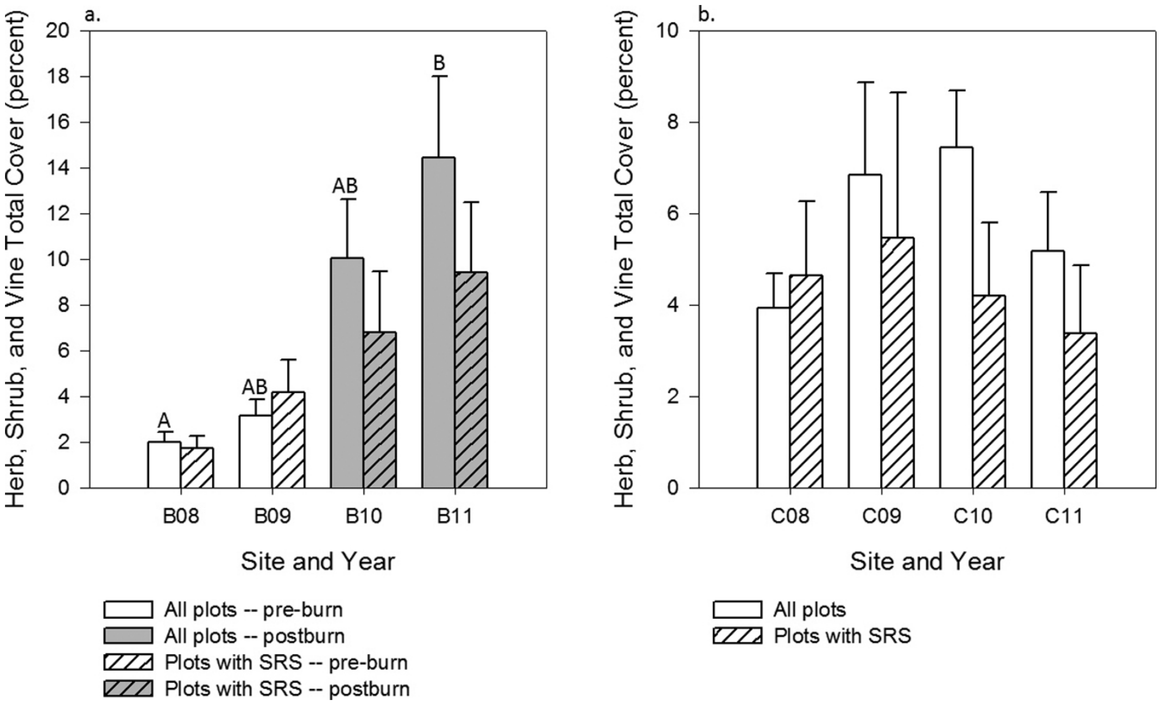

Figure 7.—Comparison, with standard errors, of total vegetation cover for all 4 years at (a) the burn site and (b) the control site. Statistical comparisons across years were made by using a generalized linear mixed model (GLIMMIX) with autoregressive covariance matrix, lognormal distribution, and an identity link function. Bars with different letters within the same graph are significantly different at P < 0.05.

At the control site, there were 35 species in 2008, 31 in 2009, 34 in 2010, and 19 in 2011. Herb, shrub, and vine diversity did not differ significantly (F = 0.62, P = 0.60) over the 4 years at the control site (Fig. 6b) though diversity declined in 2011. The same was true when evaluating only the SRS plots. Unlike the burn site, there was no increase in dominance of particular species, such as *Rubus* sp., at the control site. Indeed, *Rubus* sp. also declined in importance between 2010 and 2011 at the control site. The high levels of precipitation in March and April of 2011 (NOAA, NCDC 2014) in the area may have prevented or slowed new spring growth. The control site may have experienced more rapid rainfall than the burn site, but we have no evidence of this. Plants on the burn site could have been more likely to survive or grow, even with heavy, leaching rain, because of remnant increases in nutrients after the burn. Again, however, our data cannot confirm this.

Total Vegetation Cover

There was significantly (F = 17.82, P < 0.0001) more total understory vegetation cover for 2011 compared with both pretreatment years at the burn sites. Though not significant, there was a trend to higher cover values in the postburn years when evaluating the SRS plots (Fig. 7a). In contrast, there was no significant (F = 0.05, P = 0.99) difference in total vegetation cover at the control site across the years (Fig. 7b). Every year at the control site had less cover than the postburn years at the burn site (Fig. 7a and b). The increase in total vegetation cover at the burn site, based on the species composition information, is due to both an increase in species richness and an increase in abundance or dominance of a few species, especially in 2011. Total vegetation cover increased at the burn site but decreased at the control site between 2010 and 2011. Thus, in spite of the unmeasured variable that appeared to negatively affect SRS abundance at the control site and possibly also at the burn site between 2010 and 2011, other vegetation, especially more weedy vegetation abundance, responded positively to the burn. This positive response was sustained for an additional year at the burn site. An increase in dense understory layers, dominated by a few species, such as *Rubus* sp., *Kalmia latifolia* L., or *Dennstaedtia punctilobula* (Michx.) T. Moore, is not an uncommon response to a combined fire and canopy disturbance (Royo and Carson 2006), even though fire is often used both to reduce competing vegetation and to open the canopy for increased regeneration of certain species, such as oaks (Brose et al. 2001). Our burn site currently does not appear to be at risk of dense understory layer formation, but does show evidence of a more productive site in response to the burn, with all plant species benefitting somewhat and a few benefitting more than most.

Timing of the burn may also be a factor in the increase in productivity of the more dominant species. Although most spring burns are meant to occur before the start of the growing season, an April-May burn may have affected some early nondormant perennial plants, resulting in an increase in clonal growth. If apical meristems are nondormant, they may die in a burn, leading to the release of multiple secondary meristems and a subsequent flush of growth of competing vegetation (Platt et al. 1988). This flush of growth is most likely to occur if the burn is severe. Low- or medium-intensity spring burns do not differ in stem density or stocking after a burn from low-, medium-, or even high-intensity winter burns (Brose et al. 1999).

Based on our findings, fire may benefit SRS beyond one growing season only if it can be applied at a low enough intensity to ensure no or little increase in canopy opening. More frequent fires instead of less intense fires may help reduce competing vegetation. But if SRS is sensitive to repeat burning, they may also negatively affect SRS by directly reducing its abundance, given the annual burn that our findings suggest would be required to reduce the increase in cover of other species competing with SRS. Several fire-sensitive plants, including spring herbs (e.g., *Maianthemum racemosum* ssp.*racemosum* (L.) Link), are unable to recover after short fire intervals (Gary and Morrison 1995, Hutchinson et al. 2005). SRS is not a fire-sensitive species based on its response to one burn; however, we do not know how it would respond to multiple burns.

The decline in SRS in 2011 at the burn site, however, may be due to both an increase in competing vegetation and the unknown factor associated with SRS's decline at the control site in 2011. The decline in most species, not just SRS, in 2011 at the control site suggests that this factor may be site-specific or that positive resource changes (increased light and nutrients) related to the burn were enough for most species, especially dominant weedy species, to overcome negative factors (e.g., precipitation levels) associated with that year. Our study supports previous observations that SRS may not compete well with other plants (Dolan 2004). Thus, though there was an initial growth increase of SRS after burning, this positive response could not be sustained beyond a year.

BACI Results

The BACI results supported the GLIMMIX analyses (Table 3). Total herb, shrub, and vine cover, SRS cover, bare ground cover, and litter cover did not differ significantly in terms of pretreatment and all nonburn years. A lack of significance among the pretreatment and nonburn year data indicates that, at least before the burn, there were no site or environmental factors that influenced these variables. The contrasts of all postburn year data, including evaluating the two postburn years separately, and all nonburn years for these variables agreed with the GLIMMIX results that compared the two site types separately. Litter and total herb, shrub, and vine cover were significantly greater in both postburn years than in the pretreatment years. Bare ground cover was significantly greater in 2010 than in the pretreatment years. Herb, shrub, and vine diversity did differ significantly after the BACI analysis in terms of the pretreatment and nonburn year data. This outcome may be explained by existing species composition differences between the two sites and the decline in species richness in 2011 at the control site. Thus, the lack of an impact of the burn on plant species diversity cannot be separated from site or year-to-year environmental factors. When evaluating only the plots with SRS, there were no differences among the pretreatment years for any of the variables, including herb, shrub, and vine diversity. The absence of differences in nonburn year diversity and pretreatment data indicates that the SRS plots differed less compositionally between the control and burn sites than did all the plots together. Pretreatment and nonburn years did differ significantly from the postburn year 2010 for SRS cover when only plots with SRS were evaluated, agreeing with the GLIMMIX results.

Table 3.—BACI analysis

Contrast type	Variable	F	P-value
All plots (control and burn sites)			
Pretreatment years (2008 and 2009)	SRS cover	1.26	0.26
	Total veg. cover	1.43	0.23
	Diversity	5.92	0.016
	Bare ground cover	0.46	0.50
	Litter cover	0.19	0.66
All nonburn years (2008 and 2009 for both sites, and 2010 and 2011 for the control site)	SRS cover	1.41	0.24
	Total veg. cover	1.77	0.23
	Diversity	5.12	0.025
	Bare ground cover	0.83	0.36
	Litter cover	0.86	0.36
Postburn vs. nonburn years (2010 and 2011 for the burn site vs. 2008, 2009, 2010, and 2011 for the control site)	SRS cover	1.17	0.28
	Total veg. cover	12.58	0.0005
	Diversity	0.10	0.76
	Bare ground cover	1.03	0.31
	Litter cover	23.36	<0.0001
Postburn 2010 vs. nonburn years (2010 data for the burn site vs. 2008, 2009, 2010, and 2011 for the control site)	SRS cover	0.84	0.36
	Total veg. cover	3.62	0.059
	Diversity	0.18	0.67
	Bare ground cover	8.13	0.0049
	Litter cover	17.66	<0.0001

Contrast type	Variable	F	P-value
SRS plots only (control and burn sites)			
Pretreatment years (2008 and 2009)	SRS cover	0.35	0.56
	Total veg. cover	2.80	0.10
	Diversity	1.27	0.26
	Bare ground cover	2.37	0.13
	Litter cover	0.05	0.82
All nonburn years (2008 and 2009 for both sites, and 2010 and 2011 for the control site)	SRS cover	0.65	0.42
	Total veg. cover	1.62	0.21
	Diversity	0.47	0.50
	Bare ground cover	0.06	0.80
	Litter cover	0.82	0.37
Postburn vs. nonburn years (2010 and 2011 data for the burn site vs. 2008, 2009, 2010, and 2011 for the control site)	SRS cover	1.34	0.25
	Total veg. cover	4.84	0.032
	Diversity	0.05	0.83
	Bare ground cover	4.52	0.038
	Litter cover	3.55	0.065
Postburn 2010 vs. nonburn years (2010, data for the burn site vs. 2008, 2009, 2010, and 2011 for the control site)	SRS cover	4.67	0.035
	Total veg. cover	0.57	0.45
	Diversity	0.15	0.70
	Bare ground cover	7.43	0.0085
	Litter cover	2.04	0.16

CONCLUSIONS

Based on our results, a positive response of SRS abundance cannot be sustained for more than one growing season due to subsequent increases in other dominant plant species. The increase in other dominant plant species appeared to result from an increase in canopy opening due to overstory tree mortality after the burn. It is conceivable that a decline in SRS could occur if the prescribed burns continue to be relatively intense, leading to significant tree mortality and canopy opening. Such intense fires may be necessary on the ridge tops in order to successfully restore a savanna, but are not necessary for oak regeneration management (Schlesinger et al. 1993) on the slopes. Not burning is not an ideal option either, because SRS does benefit, at least initially, from a burn, and it is well documented that fire can promote oak regeneration (Abrams 1992, Barnes and Van Lear 1998, Brose and Van Lear 1998). It may be possible to burn the ridge tops first and then the slopes when conditions will allow for a less intense burn. Moreover, if burns are likely to be severe, we suggest that the slopes be burned earlier in the spring to reduce the risk of overlap with the growing season when apical meristems are no longer dormant (Brose et al. 1999, Platt et al. 1988). It is possible the initial positive growth response of SRS after the burn was due to the release of secondary meristems, and that SRS could not compete with other species that also benefitted from released secondary meristems.

We are concerned about the evident increase in some of the weedier species. We did not find any invasive exotic plants at either of the sites but are aware that *Alliaria petiolata* (M. Bieb.) Cavara & Grande and other invasive plants are in the vicinity. Slopes subjected to an intensive burn that opens the tree canopy will be more vulnerable to invasion, if there is a nearby propagule source (Glasgow and Matlack 2007).

Finally, information on any existing SRS seedbank and how SRS responds to fire in terms of its fitness (seed production and subsequent germination and seedling survival after a burn) will improve our ability to formulate a management strategy for this species. Repeating our SRS abundance research either at the same sites or different sites is highly recommended before deciding on a final SRS management strategy, especially given the evident year-to-year variation in SRS cover. This repeat study should include the collection of fire severity and canopy opening data so that any increases in tree mortality, canopy opening, and understory productivity can be directly related to the severity of the burn.

ACKNOWLEDGMENTS

We thank H. Smith, J. Juracko, B. Simpson, D. Fowler, and T. Jackson for their assistance with data collection. We also thank E. Byers, S. Connolly, B. Molano-Flores, V. Negron-Ortiz, J. Stanovick, and an anonymous reviewer for their willingness to provide comments.

LITERATURE CITED

Abrams, M.D. 1992. **Fire and the development of oak forests.** BioScience. 42(5): 346-353.

Barnes, T.A.; Van Lear, D.H. 1998. **Prescribed fire effects on advanced regeneration in mixed hardwood stands.** Southern Journal of Applied Forestry. 22(3): 138-142.

Bartgis, R. 1987. **Distribution and status of Arabis serotina Steele populations in West Virginia.** West Virginia Academy of Science. 59: 73-78.

Brose, P.; Schuler, T.; Van Lear, D.; Berst, J. 2001. **Bringing fire back: the changing regimes of the Appalachian mixed-oak forests.** Journal of Forestry. Nov.: 30-35.

Brose, P.T.; Van Lear, D.H. 1998. **Responses of hardwood advance regeneration to seasonal prescribed fires in oak-dominated shelterwood stands.** Canadian Journal of Forest Research. 28: 331-339.

Brose, P.T.; Van Lear, D.H.; Keyser, P.D. 1999. **A shelterwood-burn technique for regenerating productive upland oak sites in the Piedmont Region.** Southern Journal of Applied Forestry. 23(3): 158-163.

Bush, B.F. 1926. **Notes on the trees and shrubs of Missouri.** American Midland Naturalist. 10(6): 133-158.

Byers, E.A.; Vanderhorst, J.P.; Streets, B.P. 2010. **Classification and conservation assessment of upland red spruce communities in West Virginia.** Elkins, WV: West Virginia Division of Natural Resources, West Virginia Natural Heritage Program. Available at http://WVDNR.org/gov/publications/PDFFiles/RedSpruceUplands-web.PDF. (Accessed August 2, 2014).

Canham, C.D.; Denslow, J.S.; Platt, W.J.; Runkle, J.R.; Spies, T.A.; White, P.S. 1990. **Light regimes beneath closed canopies and tree-fall gaps in temperate and tropical forests.** Canadian Journal of Forest Research. 20: 620-631.

Certini, G. 2005. **Effects of fire on properties of forest soils: a review.** Oecologia. 143: 1-10.

Clarkson, R.B. 1966. **The vascular flora of the Monongahela National Forest, West Virginia.** Castanea. 31(1): 1-119.

Cooperrider, T.S.; Thorne, R.F. 1964. **The flora of Giles County, Virginia. II.** Castanea. 29(1): 46-70.

Dolan, R.W. 2004. **Conservation assessment for rock skullcap (*Scutellaria saxatilis* Riddell).** Milwaukee, WI: U.S. Department of Agriculture, Forest Service, Eastern Region. Available at http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsm91_054324.pdf. (Accessed December 22, 2014). 20 p.

Elzinga, C.L.; Salzer, D.W.; Willoughby, J.W.; Gibbs, J.P. 2001. **Monitoring plant and animal populations.** Malden, MA: Blackwell Science, Inc. 359 p.

Epling, C. 1939. **Notes on the Scutellariae of Eastern North America. I.** American Journal of Botany. 26(1): 17-24.

Gary, G.J.; Morrison, D.A. Morrison. 1995. **Effects of fire frequency on plant species composition of sandstone communities in the Sydney region: combinations of inter-fire intervals.** Australian Journal of Ecology. 20(3): 418-426.

Glasgow, L.S.; Matlack, G.R. 2007. **The effects of prescribed burning and canopy openness on establishment of two non-native plant species in a deciduous forest, southeast Ohio, USA.** Forest Ecology and Management. 238: 319-329.

Gleason, H.A.; Cronquist, A. 1993. **Manual of vascular plants of Northeastern United States and adjacent Canada.** Bronx, NY: New York Botanical Garden.

Hannah, P.R. 1987. **Regeneration methods of oaks.** Northern Journal of Applied Forestry. 4: 97-101.

Huebner, C.D. 2007. **Detection and monitoring of invasive exotic plants: a comparison of four sampling methods.** Northeastern Naturalist. 14(2): 183-206.

Hurlbert, S.H. 1984. **Pseudoreplication and the design of ecological field experiments.** Ecological Monographs. 54: 187-211.

Hutchinson, T.F.; Boerner, R.E.J.; Sutherland, S.; Kennedy Sutherland, E.; Ortt, M.; Iverson, L.R. 2005. **Prescribed fire effects on the herbaceous layer of mixed-oak forests.** Canadian Journal of Forest Research. 35: 877-890.

Interagency Taxonomic Information System (ITIS). Available at http://www.itis.gov/info.html. (Accessed August 2, 2014).

Keith, D.A. 2000. **Sampling designs, field techniques and analytical methods for systematic plant population surveys.** Ecological Management and Restoration. 1(2): 125-139.

Mazzeo, P.M. 1972. **Further notes on the flora of the Shenandoah National Park, Virginia.** Castanea. 37(3): 168-178.

Miller-Struttmann, N.E. 2013. **Rarity and reproductive biology: habitat specialists reveal a complex relationship.** Botany. 91: 349-359.

NatureServe. 2014a. **Decision tree.** Available at http://explorer.natureserve.org/decision_tree.htm. (Accessed August 2014).

NatureServe. 2014b. **NatureServe Explorer: An online encyclopedia of life [web application].** Version 7.1. Arlington, VA. Available at http://explorer.natureserve.org/servlet/NatureServe?loadTemplate=tabular_report.wmt&paging=home&save=all&sourceTemplate=reviewMiddle.wmt. (Accessed August 2014).

National Oceanic and Atmospheric Administration [NOAA], National Climatic Data Center [NCDC]. 2014. **Annual climatological summary data for Frost, WV station.** Available at http://www.ncdc.noaa.gov. (Accessed September 1, 2014).

Nott, M.P. 2006. **Shale barren rock cress (*Arabis serotina*): a literature review and analysis of vegetation data. A report to the Navy Information and Operations Command, Sugar Grove, West Virginia.** Available at www.birdpop.org/downloaddocuments/SBRC_Nott_Review.pdf. pp. 1-25. (Accessed August 2014).

Nowacki, G.J.; Abrams, M.D. 2008. **The demise of fire and "mesophication" of forests in the eastern United States.** BioScience. 58(2): 123-138.

Oksanen, L. 2001. **Logic of experiments in ecology: Is pseudoreplication a pseudoissue?** Oikos. 94: 27-38.

Olson, S.D., M.A. Homoya, E.L. Shimp. 2004. **Native plants and communities and exotic plants within the Hoosier-Shawnee Ecological Assessment Area.** Gen. Tech. Rep. NC-244. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Research Station. 267 p.

Palmer, M.E. 1987. **A critical look at rare plant monitoring in the United States.** Biological Conservation. 38: 113-127.

Paton, A. 1990. **A global taxonomic investigation of Scutellaria (Labiatae).** Kew Bulletin. 45(3): 399-450.

Platt, W.J.; Evans, G.W.; Davis, M.M. 1988. **Effects of fire season on flowering of forbs and shrubs in longleaf pine forests.** Oecologia. 76: 353-363.

Royo, A.A.; Carson, W.P. 2006. **On the formation of dense understory layers in forests worldwide: consequences and implications for forest dynamics, biodiversity, and succession.** Canadian Journal of Forest Research. 36: 1345-1362.

Schlesinger, R.C.; Sander, I.L.; Davidson, K.R. 1993. **Oak regeneration potential increased by shelterwood treatments.** Northern Journal of Applied Forestry. 10(4): 149-153.

Schwarz, C.J. 2002. **Real and quasi-experiments in capture-recapture studies.** Journal of Applied Statistics. 29(1-4): 459-473.

Strausbaugh, P.D.; Core, E.L. 1977. **Flora of West Virginia.** 2nd ed. Morgantown, WV: Seneca Books Inc.

Sun, M. 1999. **Cleistogamy in *Scutellaria indica* (Labiatae): effective mating system and population genetic structure.** Molecular Ecology. 8: 1285-1295.

Terrell, E.E. 1970. **Spring flora of the Chesapeake and Ohio Canal Area, Washington, D.C. to Seneca, Maryland.** Castanea. 35(1): 1-26.

Van Lear, D.H.; Carroll, W.D.; Kapeluck, P.R.; Johnson, R. 2005. **History and restoration of the longleaf pine-grassland ecosystem: implications for species at risk.** Forest Ecology and Management. 211: 150-165.

Walker, J.L. 1993. **Rare vascular plant taxa associated with the longleaf pine ecosystem.** Tall Timbers Fire Ecology Conference. 18: 105-126.

Wofford, B.E.; Dennis, W.M. 1976. **State records and other recent noteworthy collections of Tennessee plants.** Castanea. 41(2): 119-121.

Huebner, Cynthia D.; Karriker, Kent. 2015. **Response of smooth rock skullcap (*Scutellaria saxatilis*), a globally rare plant, to fire.** Res. Pap. NRS-28. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 16 p.

Scutellaria saxatilis Riddell (smooth rock skullcap or rock skullcap, hereafter abbreviated as SRS), a herbaceous perennial in the mint family, is a globally rare (G3) plant. In West Virginia, SRS is categorized as an S2 species (imperiled and at high risk of extinction due to a very restricted range, very few [<20] documented occurrences, or steep declines). The purpose of this study was to determine the effects of fire on SRS in West Virginia. Two forested sites (70+ years of age) within the Monongahela National Forest with no evident disturbance and with SRS populations of >1,000 individuals were selected, one in a burn area and the other in a nonburn area. Sites were sampled in early September of 2008 and 2009 (pre-burn) and 2010 and 2011 (postburn). The prescribed burn occurred in April and early May of 2010. A generalized linear mixed model with repeated measures and a spatial covariance matrix was used to determine the effects of the burn on SRS cover and associated variables including total vegetation cover, species diversity, bare ground, and litter cover. Bare ground cover increased and litter cover decreased in 2010 in response to the fire. Control and pre-burn sites did not differ significantly in terms of SRS cover over the 4-year period. The cover of SRS increased significantly in 2010 (first year postburn) compared to both pre-burn years, but decreased to pre-burn levels by 2011. Total cover of other understory vegetation increased significantly in 2010 and continued at 2010 levels in 2011 at the burn site. Thus, SRS has a temporary positive response to prescribed fire, but an increase in other ground vegetation may prevent a sustained positive response.

KEY WORDS: fire, oak community restoration, savanna restoration, rare plant conservation, rock skullcap, *Scutellaria saxatilis*

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