EXTENDED ABSTRACTS

In: Dey, Daniel C.; Stambaugh, Michael C.; Clark, Stacy L.; Schweitzer, Callie J., eds. 2012.
Proceedings of the 4th fire in eastern oak forests conference; 2011 May 17-19; Springfield, MO.
Gen. Tech. Rep. NRS-P-102. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station.

In: Dey, Daniel C.; Stambaugh, Michael C.; Clark, Stacy L.; Schweitzer, Callie J., eds. 2012.
Proceedings of the 4th fire in eastern oak forests conference; 2011 May 17-19; Springfield, MO.
Gen. Tech. Rep. NRS-P-102. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station.

GROWTH RESPONSE OF MATURE OAKS FOLLOWING TSI AND PRESCRIBED BURNING TREATMENTS

Margaret Anderson, Eric J. Holzmueller, Charles M. Ruffner, and John W. Groninger

Research Assistant (MA), Assistant Professor (EJH), Professor (CMR and JWG). Department of Forestry, 1205 Lincoln Drive – MC 4411, Southern Illinois University, Carbondale, IL 62901. EJH is corresponding author: to contact, call 618-453-3708 or email at eholzmue@siu.edu.

ABSTRACT

Oak dominated forests were historically maintained by periodic, low intensity fires in southern Illinois, but over the past century, fires have generally been suppressed, resulting in the increased abundance of fire intolerant species and decreased oak abundance (Ruffner and Groninger 2006). Recent research indicates that mechanical thinning (timber stand improvement, TSI) and prescribed fire increase the regeneration of oak species and suppress the development of fire intolerant species (Carril 2009). However, it is unclear how the residual oak trees respond to these treatments. The objective of this project was to assess how two forest management practices, TSI and prescribed fire, affect the growth of residual trees in oak dominated forests.

Three sites were selected for this study. Each site was similar in terms of slope, aspect, elevation, and site index. Prior to treatment all sites had a two-aged structure and were dominated by oak and hickory species in the overstory and sugar maple, white ash, and American elm in the understory. Each site was divided and randomly assigned one of four treatments: 1) TSI, 2) prescribed burning, 3) TSI and prescribed burning, or 4) no treatment (control). TSI treatments removed all undesirable trees < 8 inches in 2002. Stumps were treated with herbicide (Garlon 4, Triclopyr - 16 percent a.i.) immediately following cutting. Prescribed burning treatments were applied in the spring of 2002 and 2006. In 2010, we sampled 10 dominant or codominant oak trees from each treatment at each site to determine differences in growth among treatments.

Results indicated that there was no difference in growth, either pretreatment or posttreatment, among the three sites ($P > 0.12$). Mean age of the sampled trees across all sites was 102, and posttreatment growth across all treatments was minimal, ranging from 0.07 to 0.08 inches/year. There did not appear to be any increased growth following the cutting and/or burning treatments when compared to pretreatment growth ($P = 0.63$), and these treatments did not differ when compared to the control ($P = 0.58$). Growth, however, did appear to be related to tree age ($r^2 = 0.44$, $P = 0.01$) indicating that, on average, younger trees grew slightly more than older trees.

Although TSI and/or prescribed fire did not increase residual oak growth when compared to the control, these treatments can still be beneficial because of the associated increased oak regeneration (Carril 2009). Examination of the growth rates of the cored trees over time indicated these stands were mature and well beyond peak annual growth. Our results suggest that working with younger stands and/or additional management (e.g., heavier cutting) may be necessary to increase residual tree growth.

LITERATURE CITED

Carril, D.F. 2009. **Effects of repeated prescribed fire and thinning from below on understory components of southern Illinois oak-hickory forests.** Carbondale, IL: Southern Illinois University. 115 p. M.S. thesis.

Ruffner, C.M.; Groninger, J.W. 2006. **Making the case for fire in southern Illinois forests.** Journal of Forestry. 104: 78-83.

USING REMOTELY-SENSED IMAGERY TO MONITOR POST-FIRE FOREST DYNAMICS IN UPLAND OAK FORESTS ON THE CUMBERLAND PLATEAU, KENTUCKY

Mary Arthur, Christine McMichael, and Gretchen Sovkoplas

Professor (MA and GS), Department of Forestry, University of Kentucky, 214 T.P. Cooper Building, Lexington, KY 40546-0073; Associate Professor (CM). MA is corresponding author: to contact, call 859-257-2852 or email at marthur@uky.edu.

ABSTRACT

Throughout the central hardwood and southern Appalachian regions, forest managers are increasingly using prescribed fire to achieve a suite of management objectives. When such management occurs on federal lands, monitoring is mandated to determine the efficacy of prescribed fire to achieve stated objectives, yet limited funding for such efforts restricts the number, size, and spatial array of monitoring plots. Monitoring is typically done with a small number of monitoring plots across very large landscapes, and therefore, is limited in its ability to reveal the true landscape-scale impact of forest management. Based on its use in other forested ecosystems, remote sensing data have the potential to provide accurate and relatively inexpensive information in support of landscape-scale management objectives in central hardwood and southern Appalachian forests. Here we describe our preliminary efforts to develop this approach.

We used Landsat 5 Thematic Mapper satellite imagery to: (1) calculate the Normalized Difference Vegetation Index (NDVI) as a function of the red and near-infrared reflected energy, such that greener vegetation has a higher NDVI; (2) examine changes in the NDVI with burn treatment; and (3) analyze the relationships between NDVI and canopy cover measured using hemispherical photography (Alexander and others 2008). Buffered polygons were created from aerial photos to encompass each of 93 forest plots arrayed across three study sites, each with three treatments: fire-excluded, frequent burn (spring 2003, 2004, 2006, and 2008), and infrequent burn (spring 2003, 2009). The polygons were matched with plot-scale locations to capture a larger area of similar vegetation. We used Landsat 5 Thematic Mapper summer (typically June) images obtained for each year from 2002 to 2010 to calculate NDVI. Plots were located on differing landscape positions and were classified as sub-mesic, intermediate, or sub-xeric based on vegetation. Analysis of variance (ANOVA) was used to test the main effects of treatment (nested within site) and landscape position on NDVI in each year. Regression analysis was used to test the strength of the relationship between NDVI measured in one year relative to another year. Regression was also used to examine the relationship between NDVI and canopy cover.

As expected, prior to burning there was a close relationship between the NDVI in 2001 and 2002 ($R^2 = 0.66$, $p < 0.0001$). In summer 2003, after both sites were burned the previous spring, the NDVI was lower (less green) on burned compared to fire-excluded sites ($p = 0.0004$). Comparing NDVI in 2003 with 2002, the R^2 for the relationships varied from $R^2 = 0.53$ for control, $R^2 = 0.21$ for less frequent, and $R^2 = 0.17$ for frequent sites (after one burn had been conducted on both treatments in 2003). Differences in NDVI were greatest the first year after burning, despite repeated burning in subsequent years. By 2006, NDVI on less frequent burn sites was similar to that on fire-excluded

sites; both had higher NDVI compared to the frequent burn sites, which by 2006 had been burned three times, including the previous spring. A second burn in 2009 on the less frequent burn sites reduced NDVI below that of the frequent sites, but NDVI rebounded the following year.

We also examined the relationships between NDVI and canopy cover on a subset of sites (N = 33) from 2002 through 2007. In 2002, there was little variability in canopy cover or NDVI across these sites; although the relationship was significant, canopy cover explained little of the variability in NDVI ($R^2 = 0.11$; $p = 0.03$). In subsequent years, after burning the relationship between NDVI and canopy cover was strongly positive. For example, the first growing season after the first burn (2003), canopy cover explained 50 percent of the variability in NDVI ($p < 0.0001$). In 2007, after three burns in the frequent sites and one burn in the less frequent sites, there was still a strong positive relationship between canopy cover and NDVI ($R^2 = 0.38$; $p < 0.0001$), but the slope of the relationship was lower than in 2003.

The intent of the prescribed burning in our study was to remove midstory stems of fire-sensitive species with fairly small changes to the overstory. Nonetheless, immediately after the first fire, following which we found the greatest stem mortality (Arthur, unpublished data), there were highly significant differences in NDVI among treatments, and a strong relationship between NDVI and canopy cover. As overstory canopy cover was replaced by understory resprouting (Arthur, unpublished data; Blankenship and Arthur 2006; Chiang and others 2005), increasing leaf area led to increasing NDVI such that by 2010 there were small but significant differences between the control and frequent burn sites, with the less frequent sites having NDVI similar to the controls.

Future work on this project will examine additional stand characteristics (crown health, stem density, basal area) in relationship to NDVI and other spectral vegetation indices (SVIs), explore differences among landscape positions in burning effects on SVIs, and determine whether combinations of SVIs can be used to distinguish between canopy cover and understory sprouting response to burning. Prescribed fire used to create a range of habitats with more open canopies, such as oak woodlands and savannas, may lead to forest canopies that are more easily detected using this approach. Thus, we plan to expand our analysis to a landscape recently burned by wildfire which created a greater range of impacts to the forest canopy, and thus may provide a further test of this approach.

LITERATURE CITED

Alexander, H.D.; Arthur, M.A.; Loftis, D.L.; Green, S.R. 2008. **Survival and growth of upland oak and co-occurring competitor seedlings following single and repeated prescribed fires.** Forest Ecology and Management. 256: 1021-1030.

Blankenship, B.A.; Arthur, M.A. 2006. **Stand structure over nine years in burned and fire-excluded oak stands on the Cumberland Plateau, Kentucky.** Forest Ecology and Management. 225: 134-145.

Chiang, J.; Arthur, M.A.; Blankenship, B.A. 2005. **The effect of prescribed fire on gap fraction in an oak forest on the Cumberland Plateau.** Journal of the Torrey Botanical Society. 132: 432-441.

FIRESTEMII – A 2-D HEAT TRANSFER MODEL FOR SIMULATION OF STEM DAMAGE IN PRESCRIBED FIRES

Efthalia K. Chatziefstratiou, Gil Bohrer, Anthony S. Bova, and Matthew B. Dickinson

Graduate Research Associate (EKC) and Assistant Professor (GB), Department of Civil and Environmental Engineering and Geodetic Science, 416 Bolz Hall, The Ohio State University, Columbus, OH, 43210; Physical Scientist (ASB); Ecologist (MBD). EKC is corresponding author: to contact, call 614-292-9582 or email at chatziefstratiou.1@osu.edu.

Abstract.— In this work, we present FireStemII, a software tool that allows fire managers to predict tree mortality and stem injury resulting from a prescribed forest fire. FireStemII is a revision of FireStem (Jones and others 2004, 2006). When prescribing a fire, it is important to have a tool that predicts tree mortality and the extent of stem injury given the fuel conditions, fire behavior, and height aboveground along the stem. FireStemII is a physically-based thermodynamic 2-D model of tree stem injury as a function of external heat forcing. It provides increased capability for predicting fire-induced mortality and damage before a fire occurs. By directly simulating tissue temperatures, moisture loss, and charring, it determines the depth and circumferential extent of damage caused by incident heat flux around a stem at a given height. These data are further integrated to provide a depth of necrosis around the stem and an index of vascular cambium viability.

INTRODUCTION

FireStemII simulates the influence of fire as a dynamic and spatially heterogeneous heat flux boundary condition around the circumference of a virtual slice of a stem. Several other numerical models for heat transfer in tree stems subjected to fire conditions have used temperature boundary conditions (Costa and others 1991) and were typically 1-D models (Rego and Rigolot 1990), not accounting for differential heating rates and injury responses around stems. FireStem was the first, to our knowledge, to include a heat flux boundary condition (Jones and others 2004). The realism and dynamics of FireStemII will allow it to be linked to a coupled fluid dynamics and combustion (fire) model in future work. Time series of heat flux will be simulated around stems at different heights aboveground and at different locations throughout a burn area (Fig. 1). For example, as the fire-line passes a tree, turbulence on the lee side of the stem leads to increased heating and injury at the back, while the sides and front of the stem are less affected (Gutsell and Johnson 1996).

Figure 1.—FireStemII as a 2-D model of tree stem injury takes into consideration the uneven heating in the circumferential direction in case of a fire.

Stem Heating Model

The model is based on a two-dimensional heat transfer formulation:

$$\frac{\partial \rho i}{\partial t} = \frac{1}{r}\frac{\partial}{\partial r}\left(rk\frac{\partial T}{\partial r} \right) + \frac{1}{r}\frac{\partial}{\partial \theta}\left(\frac{k}{r}\frac{\partial T}{\partial \theta} \right)$$

where r is radius (cm), k is conductivity (W/(m•K)), ρ is wood density (g/cm^3), i is specific internal energy (J), T is temperature (K), and θ is angle (radians).

Stem Simulation

Stems are simulated as circular slices out of infinitely long cylinders (Jones and others 2004). The required inputs to the model include geometric information (stem diameter, outer and inner bark thickness), and physical properties (thermal conductivity, density, specific heat, and moisture content in inner bark, outer bark, and wood). Stems are divided into radial wedges (we used 16 wedges), and each wedge was divided into nodes in the radial direction. The distance between nodes is flexible and was set here to 1 mm. The new feature of FireStemII in comparison to Jones and others (2004) is that the stem is modeled as two dimensional. In addition, the numerical solver was improved with a more robust Crank-Nicholson approach, and two new routines for desiccation and bark charring were used.

Physical Phenomena

Desiccation

The change in water mass at each nodal point is calculated at each time step. The relation for water loss was taken from Morvan and Dupuy (2001):

$$\frac{\partial w}{\partial t} = W_m \frac{k_w}{\sqrt{T}} \rho \cdot M \cdot \exp\left(-\frac{E_w}{RT} \right)$$

where W_m is a multiplier that allows the water loss rate to be adjusted, M is moisture (%), T is temperature (K), ρ is wood density (g/cm^3), the coefficient k_w (6.05E5 K$^{0.5}$/s) and the exponential factor E_w/R(5956 K) are taken from Morvan and Dupuy 2001.

Bark charring

Pyrolysis is modeled in a manner analogous to water loss with the exception that in each time step, charring can occur only one node away from a previously charred node. The rate equation is based on Ragland and Aerts (1991):

$$\frac{\partial p}{\partial t} = P_m \cdot A_p \cdot \rho \cdot \left(1 - c_f\right) \cdot \exp\left(-\frac{E_p}{RT} \right)$$

where P_m is the pyrolysis multiplier, c_f is the density fraction, the coefficient A_p (7E7 s^{-1}) and the exponential factor E_p/R(15610 K) are also taken from Ragland and Aerts (1991). The combustion of charred material (glowing combustion) is not modeled.

Tissue injury

The thermally caused mortality in population of cells is described by a rate equation, where the rate of decline in tissue viability is proportional to current viability (Dickinson and Johnson 2001; Dickinson and others 2004):

$$\frac{dV}{dt} = -\kappa V(t)$$

where V is viability value between 1 (survival) and 0 (cell death), t is time (s), and κ is a species-specific, temperature-dependent rate parameter. Cellular necrosis is assumed if V is reduced below 0.5.

EXPERIMENTAL METHODS

Controlled laboratory stem heating experiments were conducted on eight regional species: red maple (*Acer rubrum* L.), sugar maple (*Acer saccharum* Marsh.), mockernut hickory (*Carya tomentosa* [Poir.] Nutt.), yellow-poplar (*Liriodendron tulipifera* L.), blackgum (*Nyssa sylvatica* Marsh.), eastern white pine (*Pinus strobus* L.), chestnut oak (*Quercus prinus* L.), and northern red oak (*Quercus rubra* L.) in order to validate FireStemII. The stem sections were fitted with three thermocouples to monitor temperatures at bark surface, beneath the bark surface, and at the cambium.

Total mass loss with heating was also determined by measuring pre- and posttreatment stem mass. Finally, depth of necrosis into the sapwood following stem heating was determined by staining stem sections with tetrazolium chloride (TTC).

RESULTS AND DISCUSSION

The laboratory stem-heating experiments of 53 tree sections of tree species mentioned above were also simulated with FireStemII. The results from the physical experiments are compared with the results from FireStemII simulations of the same cases. (Figs. 2 and 3).

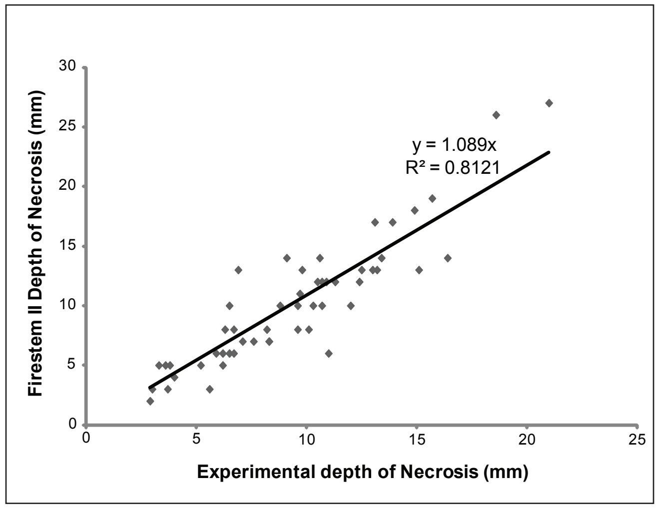

Figure 2.—Necrotic depth of 53 tree sections from eight different species. Heating experiments in the lab are compared with the necrotic depth estimated with FireStemII.

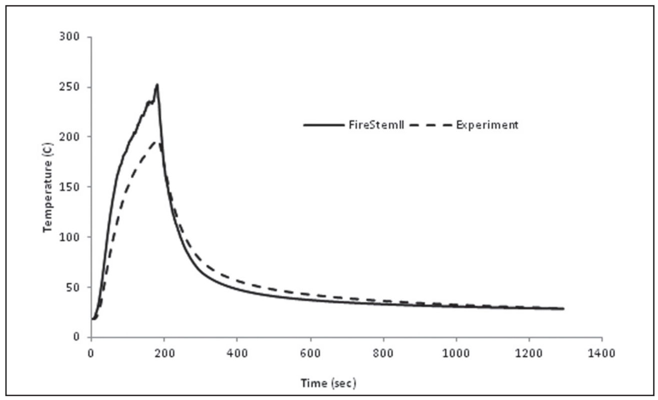

Figure 3.—Temperature variation through time just beneath bark surface in a chestnut oak tree section (stem diameter = 12.7 cm) in one lab experiment and from FireStemII simulation of the same case.

AKNOWLEDGEMENTS

The authors thank Joshua Levi Jones and Ravishankar Subramanian for collaboration and developments of earlier versions of the model. The study was funded by NASA-NESSF-Earth Sciences Fellowship #NNX09AO26 to Anthony Bova and Gil Bohrer, and through the U.S. Forest Service, Northern Research Station Agreement 09-CR-11242302-033. Bret Butler, Dan Jimenez, and Joe O'Brien provided useful feedback on an earlier draft of the manuscript.

LITERATURE CITED

Costa, J.J.; Oliveira, L.A.; Viegas, D.X.; Neto, L.P. 1991. **On the temperature distribution inside a tree under fire conditions.** International Journal of Wildland Fire. 1: 87-96.

Dickinson, M.B.; Johnson, E.A. 2001. **Fire effects on trees.** In: Johnson, E.A.; Miyanishi, K., eds. Forest fires: behavior and ecological effects. New York, NY: Academic Press: 477-525.

Dickinson, M.B.; Jolliff, J.; Bova, A.S. 2004. **Vascular cambium necrosis in forest fires: using hyperbolic temperature regimes to estimate parameters of a tissue-response model.** Australian Journal of Botany. 52: 757-763.

Gutsell, S.L.; Johnson, E.A. 1996. **How fire scars are formed: coupling a disturbance process to its ecological effect.** Canadian Journal of Forest Research. 26: 166-174.

Jones, J.L.; Webb, B.W.; Butler, B.W.; Dickinson, M.D.; Jimenez, D.; Reardon, J.; Bova, A.S. 2006. **Prediction and measurement of thermally-induced cambial tissue necrosis in tree stems.** International Journal of Wildland Fire. 15: 3-17.

Jones, J.L.; Webb, B.W.; Jimenez, D.; Reardon, J.; Butler, B. 2004 **Development of an advanced one-dimensional stem heating model for application in surface fires.** Canadian Journal of Forest Research. 34: 20-30.

Morvan, D.; Dupuy; J.L. 2001. **Modeling of fire spread through a forest fuel bed using a multiphase formulation.** Combustion and Flame. 127: 1981-1994.

Ragland, K.W.; Aerts, D.J. 1991. **Properties of wood for combustion analysis.** Biresource Technology. 37: 161-168.

Rego, F.; Rigolot, E. 1990. **Heat transfer through bark - a simple predictive model.** In: Goldammer, J.G.; Jenkins, M.J., eds. The Hague, Netherlands: SPB Academic Publishing: 157-162.

EFFECTS OF TWO PRESCRIBED FIRES ON RED MAPLE REGENERATION ACROSS FOUR LEVELS OF CANOPY COVER IN NORTHERN LOWER MICHIGAN

William W. DeBord, David S. Buckley, Terry L. Sharik, Jason P. Hartman, and Jason G. Henning

Graduate Research Assistant (WWD) and Associate Professor (DSB and JGH),
Department of Forestry, Wildlife and Fisheries, 274 Ellington Plant Sciences Building,
University of Tennessee, Knoxville, TN 37996-4563; Professor (TLS),
Department of Wildland Resources, Utah State University; Unit Manager (JPH),
Michigan Department of Natural Resources and Environment.
DSB is corresponding author: to contact, call 865-974-7978 or email at dbuckley@utk.edu.

INTRODUCTION

Mixed oak forests persist as a predominant forest type in eastern U.S. forests, but very little oak recruitment into the overstory has occurred since the implementation of fire suppression in the early 1900s (Abrams 1992). Advanced oak regeneration on high quality and some intermediate quality sites is being replaced in the midstory by more mesic, shade tolerant species (Abrams and Nowacki 1992). A large proportion of the total increase in the density of competitors can often be attributed to an influx of red maple (*Acer rubrum* L.) (Abrams 1998, 2005). Current thinking is that reintroduction of representative fire regimes into eastern oak systems may help promote oak recruitment and impede the development of red maple competition. It has been suggested that oak seedlings and saplings are better adapted to fire than those of red maple and other competitors due to greater allocation of resources to roots and several other adaptations (Brose and Van Lear 1998). As a result, it follows that multiple fires should place red maple and other competitors at an ever-increasing disadvantage relative to oak species.

As a component of a larger, long-term oak regeneration project established in 1990, the objectives of this study were to determine the additive effects of two spring prescribed fires on red maple regeneration across four levels of canopy cover, and test the hypothesis that red maple densities in medium (25 cm tall – 2.54 cm diameter at breast height [d.b.h.]) and large (>2.54 cm d.b.h.) size classes would be reduced to a greater degree following the second fire than following the first.

METHODS

Second-growth natural oak stands on state forest land in northern Lower Michigan were selected as study sites. The sites are located within the Grayling Outwash Plain of the Highplains District of the northern Lower Peninsula (Albert 1995). All stands occurred on sandy, mixed, frigid, Alfic Haplorthods developed in pitted outwash. Soils in all stands had comparable physical and chemical properties (Kim and others 1996). All sites were moderately productive with slopes ≤5 percent. Site index for northern red oak (*Quercus rubra* L.) was 17.8 m at 50 years (Carmean 1978). Based on counts of annual growth rings on stumps following overstory treatment implementation in 1991, the stands studied were 106-118 years old in 2009.

Each of three 1.74 ha replicate oak blocks was subdivided into four overstory plots measuring 66 × 66 m (0.44 ha). Each overstory plot was randomly assigned one of four canopy cover treatments: clearcut (CC), 25 percent residual canopy cover (25 percent), 75 percent residual canopy cover (75 percent), or uncut control (UC). All stands were cut between fall 1990 and early spring 1991. Four understory treatment subplots measuring 15 × 15 m (0.02 ha each) were arranged in a square at the center of each canopy treatment plot in 1991, and the understory treatments were maintained intermittently through 1996.

All stands were burned on May 15, 2002 and burned again on May 16, 2008. The goal in both years was to achieve strip-head fires with 0.9 m flame lengths in order to top kill all hardwood seedlings on all sites. Maximum fire temperatures were documented with temperature indicating paints (B.J. Wolfe Enterprises Inc., Agoura Hills, CA) painted on the bottom surface of horizontally oriented ceramic tiles. One ceramic tile was mounted on a steel stake 0.6 m above the soil surface in the center of each of the four understory plots within each overstory treatment plot.

Natural regeneration was measured in 2001, 2003, 2006, and 2009 at four sampling points within each understory plot near the end of the growing season in late July/early August. Genets of woody stems were tallied by species into three size classes: 1) Small (stems <25 cm height); 2) Medium (stems 25 cm height – 2.54 cm diameter); and 3) Large (stems 2.54 cm diameter – 10 cm diameter). Small size class stems were measured within 1 m^2 quadrats. Medium and large size class stems were measured within 2 and 4 m diameter circular plots, respectively. Smaller sampling plots were nested within larger plots, and all three plots were centered on each of the four sampling points within each of the understory plots.

Paired t-tests were used to evaluate differences between the differences in pre- and postburn seedling densities associated with each fire using SAS/STAT® software (version 9.2, SAS Institute Inc., Cary, NC). Hypothesis tests were performed with $\propto = .05$.

RESULTS AND DISCUSSION

Maximum fire temperatures ranged from 40 to 100 °C at 0.6 m above the soil surface. Temperatures were comparable in both years, and flame lengths were typically 0.3 m or less. In the case of medium and large red maple stems, in most of the overstory treatments there were differences between the differences in maple density before and after Fire 1 and the differences in density before and after Fire 2 (Table 1). Differences associated with the second fire were far smaller in most cases, however, than those associated with the first fire (Table 1). Consequently, the hypothesis that red maple stems in the two larger size classes would be reduced more by the second fire than by the first was not supported.

Many of the larger surviving red maple stems were stump sprouts that emerged from thinned midstory saplings following implementation of canopy treatments in 1991. These sprouts were able to develop over 11 growing seasons prior to the first prescribed fire. Densities may have been reduced to a greater degree following the first fire due to high mortality among stressed, ill-formed, or diseased stems, leaving only the most robust, well-established stems which then had six growing seasons to recover before the second fire. Reducing these robust stems may require continued repeated burning.

Prescribed fires implemented more frequently than in this study may help to reduce the density of large red maple stems that are able to survive and recover over longer time periods between fires. More frequent burns, however, will also have effects on oak stems. Overall, the two prescribed fires had a cumulative effect, reducing red maple densities by 46-74 percent in the medium size class and by 53-87 percent in the large size class, which may improve the future competitive position of oak seedlings and saplings.

ACKNOWLEDGMENTS

This research was funded by the University of Tennessee Department of Forestry, Wildlife and Fisheries, and the Michigan Department of Natural Resources (DNR). Special thanks to several Michigan DNR personnel and to Chris Miller, John Johnson, and Stephen Grayson for their help in the field.

Table 1.—Red maple densities by size class, year, and canopy treatment, differences between pre- and postfire densities associated with each prescribed fire, and results for paired t-tests conducted within canopy treatments to compare differences in red maple density associated with each fire.

Medium Size Class

Canopy Treatment[1]	2001 Density	2003 Density	Fire 1 Difference	2006 P-value[2]	2009 Density	Fire 2 Density	Difference
	Stems/ha	*Stems/ha*	*Stems/ha*		*Stems/ha*	*Stems/ha*	*Stems/ha*
CC	9416.68	4244.14	–5172.54	0.0605	4111.51	2453.64	–1657.87
25%	23475.37	13262.92	–10212.45	**0.0094**	14920.79	10941.91	–3978.88
75%	22480.65	12599.78	–9880.88	**0.0057**	17772.32	12135.57	–636.74
UC	16114.45	6432.52	–9681.93	**0.0063**	8289.33	4708.34	–3580.99

Large Size Class

Canopy Treatment	2001 Density	2003 Density	Fire 1 Difference	2006 P-value	2009 Density	Fire 2 Density	Difference
	Stems/ha	*Stems/ha*	*Stems/ha*		*Stems/ha*	*Stems/ha*	*Stems/ha*
CC	2287.85	1177.08	–1110.77	**0.0082**	1110.77	1077.61	–33.16
25%	1823.65	629.99	–1193.66	**0.0035**	613.41	464.20	–149.21
75%	646.57	132.63	–513.94	**0.0388**	198.94	82.89	–116.05
UC	116.05	16.58	–99.47	0.3400	66.31	16.58	–49.74

[1]CC = clearcut; 25 percent = 25 percent canopy cover; 75 percent = 75 percent canopy cover; and UC = uncut controls.
[2]Bold P-values indicate statistically significant differences between differences in red maple density before and after Fire 1 and differences in red maple density before and after Fire 2 suggested by paired t-tests with alpha = 0.05.

LITERATURE CITED

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

Abrams, M.D. 1998. **The red maple paradox. BioScience.** 48: 355-364.

Abrams, M.D. 2005. **Prescribing fire in eastern oak forests: Is time running out?** Northern Journal of Applied Forestry. 22: 190-196.

Abrams, M.D.; Nowacki, G.J. 1992. **Historical variation in fire, oak recruitment, and post-logging accelerated succession in central Pennsylvania.** Bulletin of the Torrey Botanical Club. 119: 19-28.

Albert, D.A. 1995. **Regional landscape ecosystems of Michigan, Wisconsin, and Minnesota: a working map and classification.** Gen. Tech. Rep. NC-178. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experimental Station. 250 p.

Brose, P.H.; 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.

Carmean, W.H. 1978. **Site index curves for northern hardwoods in northern Wisconsin and Upper Michigan.** Res. Pap. NC-160. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experimental Station. 16 p.

Kim, C.; Sharik, T.L.; Jurgensen, M.F.; Dickson, R.E.; Buckley, D.S. 1996. **Effects of nitrogen availability on northern red oak seedling growth in oak and pine stands in northern lower Michigan.** Canadian Journal of Forest Research. 26: 1103-1111.

TALL LARKSPUR BENEFITS FROM FIRE IN CHINKAPIN OAK WOODLANDS

Dan Drees

Fire Ecologist, National Park Service, Ozark National Scenic Riverways, Van Buren, MO 63965.
To contact, call 573-323-8234 ext. 26 or email at daniel_drees@nps.gov.

ABSTRACT

Tall larkspur (*Delphinium exaltatum*) is a globally rare wildflower (NatureServe 2010). In Missouri, tall larkspur is strongly associated with chinkapin oak (*Quercus muehlenbergii* Engelm.) dominated woodlands (85 percent of sites) (Mullarkey and others 2008). It is an important oak woodland indicator species that requires strong filtered sunlight to maintain viable populations. In the absence of fire, the woodlands that tall larkspur prefers are subject to succession towards forest. The subsequent increase in shade greatly reduces tall larkspur populations (Fig. 1).

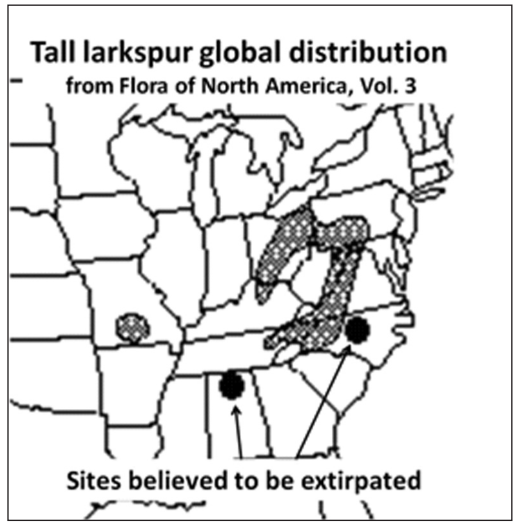

Figure 1.

According to www.natureserve.org, “A primary threat to tall larkspur is loss of habitat due to succession of vegetation in the absence of a natural fire regime. Encroachment of trees and shrubs (e.g., eastern redcedar [*Juniperus virginiana* L.]) into occupied habitat has likely resulted in the loss of many individuals and populations over time.” Historically, fire kept woody plants from becoming too dense in tall larkspur’s preferred habitat. This attractive wildflower may persist naturally without fire for decades on some sunny, steep slopes and the sunny banks of small high-gradient streams. This is especially true if windstorms, ice storms, or selective timber harvesting have thinned the tree canopy. However, only fire will prevent leaf litter accumulation from suppressing tall larkspur seedling germination over most of its preferred habitat.

Tall larkspur population inventories at Ozark National Scenic Riverways (ONSR) between 1984 and 2008 documented dramatic population declines at unburned sites. The largest population (650 plants) declined by 64 percent in total number and by 94 percent in number of reproductive plants. In 2009, the Ozark Highlands fire ecology crew of the National Park Service discovered a new population of 2,481 tall larkspur plants in a fire managed woodland at ONSR. It is the largest reported natural population of tall larkspur in a prescribed fire management unit anywhere.

In 2010, ONSR embarked on an ongoing project with permanent plots to study the fire ecology of tall larkspur in the Current River watershed where 15 of 17 populations in Missouri are located. We would like to collaborate with managers of natural tall larkspur populations throughout the plant’s range.

LITERATURE CITED

Flora of North America Editorial Committee. 1997. **Flora of North America north of Mexico. Volume 3: Magnoliophyta: Magnoliidae and Hamamelidae.** New York, NY: Oxford University Press. 616 p.

Mullarkey, A.; Farrington, S.; Drees, D. 2008. **Habitat, population status and management of tall larkspur (*Delphinium exaltatum*) in Missouri.** [Poster]. In: Missouri natural resources conference; 2008 Jan 30 - Feb 1; Camdenton, MO.

NatureServe. 2010. **NatureServe explorer: an online encyclopedia of life.** [Web application]. Version 7.1. NatureServe: Arlington, Virginia. Available at http://www.natureserve.org/explorer. (Accessed 2011 May 12).

LOCAL COLLARED LIZARDS NEEDED LANDSCAPE-SCALE FIRE

Dan Drees

Fire Ecologist, National Park Service, Ozark National Scenic Riverways, Van Buren, MO 63965.
To contact, call 573-323-8234 ext. 26 or email at daniel_drees@nps.gov.

ABSTRACT

Evolutionary biologist Dr. Alan Templeton used 15 years of field research on eastern collared lizards to advocate for landscaped-scale prescribed burn units beginning in 1994. Another 18 years of field research by Dr. Templeton and his colleagues demonstrated that landscape-scale burns are necessary for a variety of reasons. In addition to restoring glade habitat for collared lizards, the large burns were necessary to open surrounding woodlands as travel corridors (Templeton and others 2007). Burning the surrounding woodlands permitted additional glades to be colonized and facilitated genetic dispersal.

Dr. Templeton started in 1979 with a search of the Stegall Mountain vicinity (Current River watershed in Missouri) to determine if historic populations of the collared lizard still existed. No collared lizards were observed in the initial surveys. The areas numerous igneous and dolomite glades had become too overgrown in trees, especially eastern redcedar. The historic populations of collared lizards probably died out due to shading of their open glade habitat.

In 1982, cedar removal and small prescribed fires were initiated to reopen the best remaining glades on nearby Stegall Mountain. In 1984, collared lizards were captured from several healthy populations 40 to 80 miles away and released on Stegall Mountain. After a few years, Dr. Templeton realized that collared lizards were not moving to nearby glades that were separated by thin bands of dense woody vegetation, some only 50 meters wide.

Dr. Templeton, as part of a Biodiversity Task Force commissioned by the Missouri Department of Conservation, suggested burning several hundred acres of glades and woodlands at once. At the time, this was a radical concept in Missouri. In 1994, the first landscape-scale burn was completed on Stegall Mountain. The collared lizards immediately responded by colonizing several nearby glades.

One of Dr. Templeton's students found that grasshoppers were the primary food of local collared lizards (Östman and others 2007). Another student discovered that there was a 650 percent increase in grasshoppers in the prescribed burn units (Jon Hallemeier and A. Templeton, unpublished results). Grasshoppers and other ground dwelling insects are also the primary food for turkey hatchlings and many species of grassland birds.

Indeed, the National Wild Turkey Federation (NWTF) has recognized for a long time that glades and their surrounding woodlands should be properly managed with prescribed fire to create the optimal mix of nesting cover, brood rearing habitat, and acorn production. Locally, the NWTF has been an important ally in promoting the restoration of fire-dependent habitats that support a diversity of rare plants and animals in addition to important game species.

Dr. Templeton's dream was to see at least 2,000 contiguous acres in prescribed fire management. In 2009, the National Park Service, the Missouri Department of Conservation, and The Nature Conservancy joined forces to complete over 5,000 contiguous acres of prescribed burns in the Stegall Mountain area in just one season.

Collared lizards have now colonized all of the suitable glades within these adjoining burn units. Most importantly, the collared lizards are indicators that landscape-scale prescribed fire has restored, stabilized, or increased populations of many other fire-dependent conservative species.

LITERATURE CITED

Östman, O.; Griffin, N.W.; Strasburg, J.L.; Brisson, J.A.; Templeton, A.R.; Knight, T.M.; Chase, J.M. 2007. **Habitat area affects arthropod communities directly and indirectly through top predators.** Ecography. 30: 359-366.

Templeton, A.R.; Neuwald, J.L.; Brazeal, H.; Robertson, R.J. 2007. **Restoring demographic processes in translocated populations: the case of collared lizards in the Missouri Ozarks using prescribed forest fires.** Israel Journal of Ecology and Evolution. 53: 179-196.

EFFECTS OF WILDFIRES AND LIMING OF PINE-OAK-HEATH COMMUNITIES IN THE LINVILLE GORGE WILDERNESS, WESTERN NORTH CAROLINA

Katherine J. Elliott, James M. Vose, Jennifer D. Knoepp, and William Jackson

Research Ecologist (KJE), Research Ecologist and Project Leader (JMV), and Research Soil Scientist (JDK), U.S. Forest Service, Southern Research Station, Coweeta Hydrologic Laboratory, Otto, NC 28763; Air Quality Specialist (WJ).
KJE is corresponding author: to contact, call 828-524-2128 or email at kelliott@fs.fed.us.

ABSTRACT

Linville Gorge Wilderness (LGW) is a Class I area in the southern Appalachian Mountains, western North Carolina. Over the last 150 years, LGW has been subject to several wildfires, varying in intensity and extent (Newell and Peet 1995). In November 2000, a wildfire burned 4000 ha in the wilderness; the fire ranged in severity across the northern portion of the wilderness from low severity in coves to high severity along ridges and bluffs (Reilly and others 2006). In May 2007, the Pinnacle wildfire burned ca. 2000 ha in the southern portion of the wilderness. A large portion of the Pinnacle fire overlapped the area previously burned in 2000, resulting in much of the central section of LGW being burned twice in less than 7 years. In addition, dolomitic lime was aerially applied at a rate of 1120 kg ha^{-1} on the most severely burned area. We hypothesized that liming the most severely burned area would accelerate the restoration of acidic, nutrient depleted soils by adding basic cations, balancing soil pH, reducing soil and soil solution Al, and subsequently increase ecosystem productivity. We sampled vegetation (composition, biomass, and foliar nutrients) and soil nutrients in five treatment areas in LGW over a 2-year period following the most recent wildfire. The treatment areas were: severely burned twice (2000 and 2007) plus dolomitic lime application (2xSBL); moderately burned twice plus lime (2xMBL); severely burned twice, no lime (2xSB); moderately burned once (2000), no lime (1xMB); and an unburned and unlimed reference area (REF).

All wildfire burned sites experienced overstory mortality (>300 stems ha^{-1}). There were no live overstory trees on 2xSBL, and 2xSB had a larger number of dead trees than all other sites. The large number of dead pines on all sites, including the reference, was due in large part to a southern pine beetle (*Dendroctonus frontalis* Zimm.) outbreak in 1999-2001. The three areas impacted by the recent wildfire had little to no live pine in the overstory. Overstory biomass was lower on the severely burned areas (2xSBL and 2xSB) than the other treatments, with no significant change between 2008 and 2009 (Fig. 1). By 2009, understory density was higher on 2xSBL than the other treatments (Fig. 2a), with higher numbers of tree species (Fig. 2b). We found no differences in shrub density among burned treatments, and only 2xSB had higher shrub density than REF (Fig. 2c). As expected, ericaceous (heath) species sprouted after fire, and their density increased between 2008 and 2009 for all recently burned treatments.

We found no differences in foliar nitrogen (N) concentration among treatments for either evergreen or deciduous species (Table 1). Evergreen foliar calcium (Ca) was greater on 2xMBL and REF than the severely burned sites (2xSBL and 2xSB); whereas deciduous foliar Ca and phosphorus (P) were greater on 2xMBL than 2xSBL, 1xMB and REF (Table 1).

With few exceptions, we found significant date, treatment, date x treatment interaction effects for soil exchangeable cations for both shallow (0-10 cm

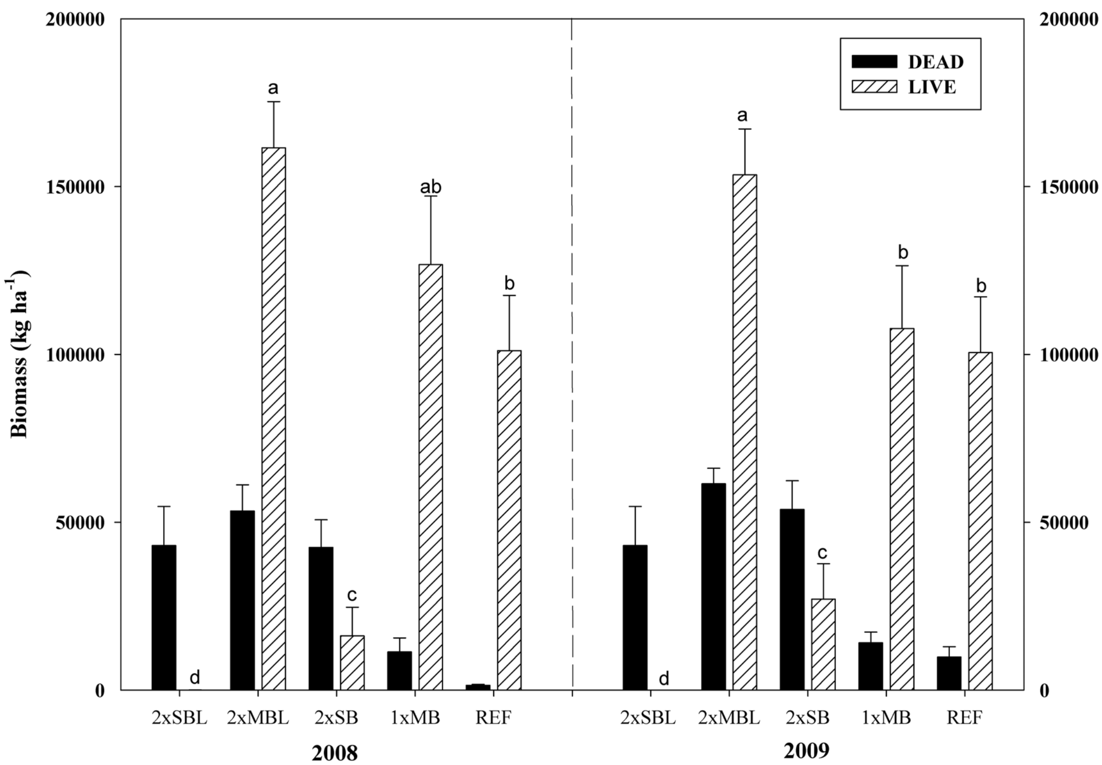

Figure 1.—Overstory (≥ 5.0 cm diameter at breast height [d.b.h.]) biomass with standard error bars of live and standing dead trees for the five treatment areas the first (2008) and second (2009) growing seasons after wildfire in Linville Gorge Wilderness, western North Carolina. The treatments were severely burned twice (2000 & 2007) plus dolomitic lime application (2xSBL); moderately burned twice with lime (2xMBL); severely burned twice, no lime (2xSB); moderately burned once (2002), no lime (1xMB); and an unburned and unlimed reference area (REF). Bars with different letters denote significant (α≤0.05) differences among treatments within a year.

depth) and deep (10-30 cm depth) soils (Table 2). Soil exchangeable Ca^{2+} had a significant treatment effect, but no date or date x treatment interaction effects (Table 2). Soil Ca^{2+} was greater on 2xMBL and 2xSBL than 2xSB, 1xMB, and REF for shallow soil, and greater on 2xMBL than 2xSBL for deep soil (Table 2). Soil extractable aluminum (Al^{3+}) was less on 2xSBL than 2xSB, 1xMB, and REF, which resulted in higher Ca/Al ratios on 2xSBL and 2xMBL than 2xSB, 1xMB, and REF.

Wildfires are landscape-scale disturbances and have the potential to significantly impact biogeochemical processes by altering pools and fluxes of carbon and nutrients (Debano and others 1998, Knoepp and others 2005). The magnitude and duration of these responses depends on the interactions among preburn conditions, burn severity, postfire precipitation regime, topography, soil characteristics, and vegetative recovery rate (Robichaud 2005). We expected that the effects of the most recent wildfire and lime application would be observed over a 2-year period, possibly longer, as Ca and Mg leach into the soil where it can be taken up by the recovering vegetation (Elliott and others 2002, Knoepp and Swank 1997). In our study, the moderately burned site with lime application had the highest soil and foliar Ca suggesting that the lime application was subsequently taken up by vegetation.

In contrast, the severely burned site with lime had greater soil Ca but lower foliar Ca than the moderately burned site even though both sites received lime application. Soil Al was lowest on 2xSBL, suggesting that the lime addition did improve the soil Ca/Al ratio since it was higher on this site than the others. Al^{3+} increased and Ca/Al ratio decreased over time for both the shallow and deep soil depths (Table 2). Thus, the lime addition did improve the soil Ca/Al ratio, but the response was transitory. Soil Ca/Al ratios remain well below the toxicity threshold <1.0 (Cronon and Grigal 1995) on all treatment areas in LGW.

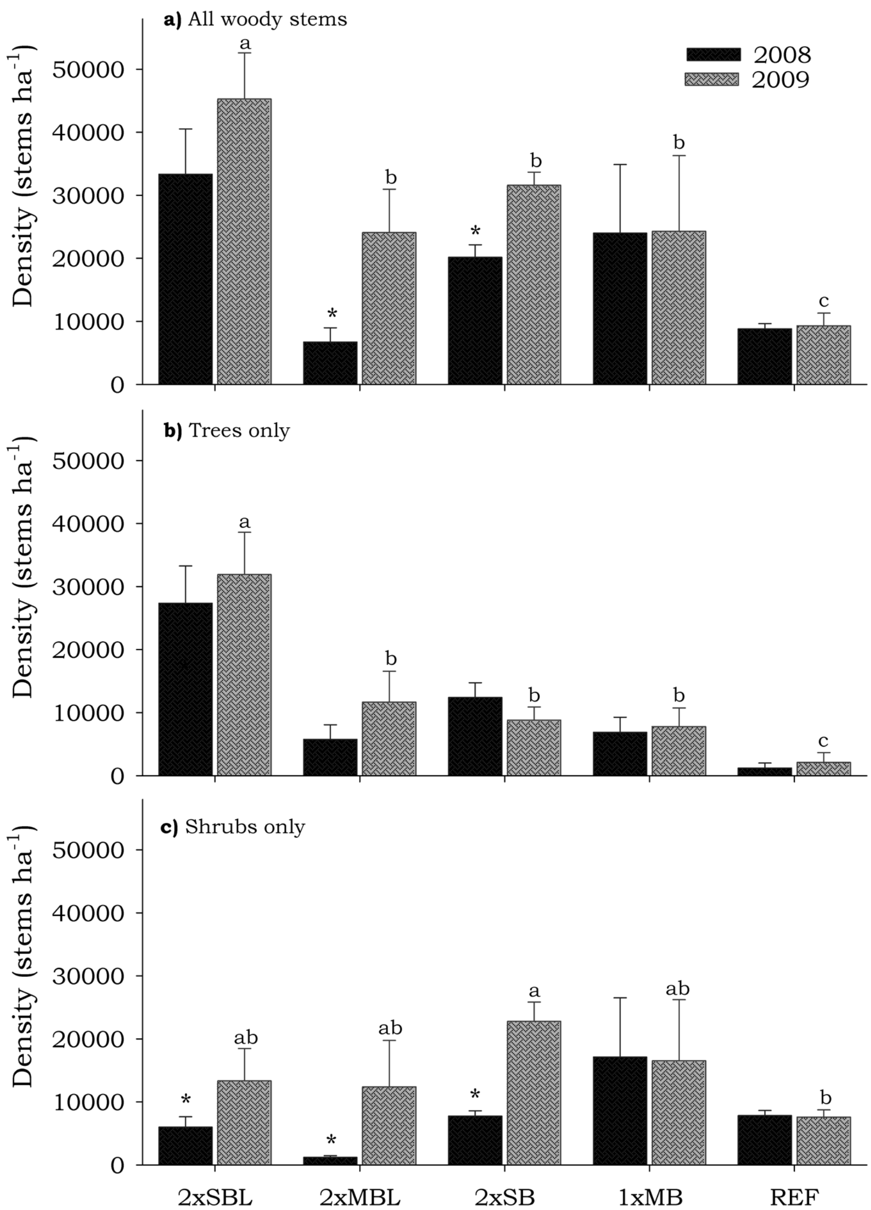

Figure 2.—Understory (shrubs and trees <5.0 cm d.b.h.) density with standard error bars: a) all woody species, b) trees only, and c)shrubs only for the five treatment areas the first (2008) and second (2009) growing seasons after wildfire in Linville Gorge Wilderness, western North Carolina. Treatment labels are the same as in Figure 1. Bars with different letters denote significant (α≤0.05) differences among treatments within a year. Asterisk denotes significant (α≤0.05) difference between years within a treatment.

Table 1.—Nutrient concentrations (with standard errors in parenthesis) of evergreen and deciduous leaf tissue for the five treatment areas: severely burned twice (2002 and 2007) plus dolomitic lime application (2xSBL); moderately burned twice plus lime (2xMBL); severely burned twice, no lime (2xSB); moderately burned once (2002), no lime (1xMB); and an unburned and unlimed reference area (REF).

	2xSBL	2xMBL	2xSB	1xMB	REF
N (percent)					
Evergreen	0.779 (0.032)	0.961 (0.021)	0.877 (0.032)	0.961 (0.064)	0.923 (0.052)
Deciduous	1.696 (0.067)	1.775 (0.132)	1.553 (0.059)	1.639 (0.057)	1.712 (0.042)
Ca ($\mu g\ g^{-1}$)					
Evergreen	6687 b (373)	10214 a (1001)	7180 b (313)	8878 ab (646)	10163 a (506)
Deciduous	4927 b (470)	6697 a (158)	5503 ab (560)	5164 b (510)	5014 b (556)
P ($\mu g\ g^{-1}$)					
Evergreen	736 (33)	780 (18)	707 (28)	670 (48)	739 (68)
Deciduous	1506 ab (36)	1612 a (53)	1445 ab (42)	1249 c (29)	1357 bc (24)
Al ($\mu g\ g^{-1}$)					
Evergreen	52 (6)	59 (5)	47 (4)	169 (65)	193 (92)
Deciduous	91 b (30)	104 ab (21)	138 ab (9)	166 a (28)	84 b (11)

Notes: Values followed by different letters are significantly different ($p < 0.05$) among treatments (SAS 2002-2003).

Table 2.—Mean soil chemistry (with standard errors in parenthesis) for the five treatment areas in the Linville Gorge Wilderness, western North Carolina, USA. Treatment labels are the same as in Table 1. All values are in $cmol_c\ kg^{-1}$ except for Ca/Al molar ratio.

	2xSBL	2xMBL	2xSB	1xMB	REF
0-10 cm soil depth					
Nitrogen	0.107 b (0.021)	0.164 a (0.011)	0.143 ab (0.012)	0.145 ab (0.005)	0.139 ab (0.005)
NO_3–N	0.0009 (0.0004)	0.0003 (0.0003)	ND†	ND	ND
NH_{4+}-N	0.118 (0.021)	0.105 (0.022)	0.070 (0.005)	0.104 (0.011)	0.072 (0.005)
HPO_4^{2-}	0.236 ab (0.009)	0.250 a (0.015)	0.195 b (0.008)	0.241 ab (0.014)	0.251 a (0.010)
Ca^{2+}	7.322 ab (0.502)	9.616 a (1.662)	1.854 c (0.363)	4.009 b (0.630)	0.626 c (0.059)
Al^{3+}	76.36 c (6.234)	167.55 a (8.232)	124.00 b (6.404)	110.09 b (4.752)	109.40 b (7.700)
Ca/Al	0.102 a (0.006)	0.072 ab (0.017)	0.015 c (0.003)	0.037 bc (0.006)	0.006 c (0.001)
10-30 cm soil depth					
Nitrogen	0.046 b (0.009)	0.070 a (0.004)	0.078 a (0.006)	0.062 ab (0.003)	0.060 ab (0.003)
NO_3–N	ND	ND	ND	ND	ND
NH_{4+}-N	0.040 b (0.005)	0.044 b (0.003)	0.052 a (0.003)	0.053 a (0.003)	0.038 b (0.002)
HPO_4^{2-}	0.111 b (0.013)	0.142 ab (0.005)	0.143ab (0.008)	0.136 ab (0.009)	0.162 a (0.014)
Ca^{2+}	0.956 b (0.038)	1.519 a (0.251)	0.574 bc (0.146)	0.606 bc (0.063)	0.218 c (0.015)
Al^{3+}	57.83 c (4.982)	116.57 a (4.868)	87.55 b (6.530)	76.42 bc (3.719)	81.11 b (2.747)
Ca/Al	0.018 a (0.001)	0.016 a (0.003)	0.006 bc (0.001)	0.009 b (0.001)	0.003 c (0.0002)

†ND = samples below the detection limits of our methods. Values within a soil depth followed by different letters are significantly different ($p < 0.05$) among sites (SAS 2002-2003). Values were averaged across time; and then, mean values and standard errors were based on the five plots per treatment area ($n = 5$).

LITERATURE CITED

Cronon, C.S.; Grigal, D.F. 1995. **Use of calcium/ aluminum ratios as indicators of stress in forest ecosystems.** Journal of Environmental Quality. 24(2): 209-226.

DeBano, L.F.; Neary, D.G.; Folliott, P.F. 1998. **Fire's effects on ecosystems.** New York: John Wiley & Sons, Inc. 333 p.

Elliott, K.J.; Boring, L.R.; Swank, W.T. 2002. **Aboveground biomass and nutrient pools in a Southern Appalachian watershed 20 years after clearcutting.** Canadian Journal of Forest Research. 32(4): 667-683.

Knoepp, J.D.; Swank, W.T. 1997. **Long-term effects of commercial sawlog harvest on soil cation concentrations.** Forest Ecology and Management. 93(1-2): 1-7.

Knoepp, J.D.; DeBano, L.F.; Neary, D.G. 2005. **Soil chemistry.** In: Neary, D.G.; Ryan, K.C.; DeBano, L.F., eds. Wildland fire in ecosystems: effects of fire on soil and water. Gen. Tech. Rep. RMRS-42. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 53-71.

Newell, C.L.; Peet, R.K. 1995. **Vegetation of Linville Gorge Wilderness, North Carolina.** Curriculum in Ecology and Department of Biology. Chapel Hill, NC: University of North Carolina.

Reilly, M.J.; Wimberly, M.C.; Newell, C.L. 2006. **Wildfire effects on plant species richness at multiple spatial scales in forest communities of the southern Appalachians.** Journal of Ecology. 94(1): 118-130.

Robichaud, P.R. 2005. **Measurement of post-fire hillslope erosion to evaluate and model rehabilitation treatment effectiveness and recovery.** International Journal of Wildland Fire. 14(4): 475-485.

SAS Institute Inc. 2002-2003. **SAS/STAT Guide for personal computers.** Version 9.1, Cary, NC.

COMMUNITIES AND DENSITY ESTIMATES OF HISTORICAL AND CURRENT MISSOURI OZARKS FORESTS

Brice B. Hanberry, Hong S. He, Daniel C. Dey, John M. Kabrick, and Brian J. Palik

Research Associate (BBH) and Associate Professor (HSH), Department of Forestry, 203 Natural Resource Building, University of Missouri, Columbia, MO, 65211; Research Forester (DCD, JMK, BJP). BBH is corresponding author: to contact, call 573-875-5341 ext. 230 or email at hanberryb@missouri.edu.

INTRODUCTION

Forests in Missouri changed after European settlement, due in part to timber harvest, fire suppression, grazing, and conversion to agricultural and urban land use. There are few current examples of fire-mediated composition and structure present in oak and pine savannas and woodlands. Reconstruction of historical landscapes can provide a reference for restoration efforts.

METHODS

The United States General Land Office (GLO) surveyed trees during 1815-1864 in the Missouri Ozarks. The U.S. Forest Service's Forest Inventory and Analysis (FIA) program surveys current forests. Using data from 2004-2008, we selected live trees with a diameter at breast height (d.b.h) ≥ 12.7 cm. Our ecological unit was a subsection divided into land types (ECOMAP 1993).

For community assignment in an ecological unit, the number of trees had to be ≥200 per ecological unit, and percent composition of species had to be ≥10 per ecological unit to be a dominant species in the community. Community order was based on descending mean percent composition for all GLO trees.

For density estimation in an ecological unit, FIA plots had to be 100 percent forest land. For historical forests, we used the Morisita plotless density estimator and applied corrections for surveyor bias. For current forests, we calculated trees per acre using the expansion factor of 6.018046, based on one tree representing the inverse of the plot area in acres (i.e., $1/[4*0.04154172]$). To predict densities continuously for 835,000 Soil Survey Geographic (SSURGO) Database polygons (mean area of 10 ha), we used calculated plot density, 24 soil and topographic predictor variables, and random forests, an ensemble regression tree method.

RESULTS AND DISCUSSION

Across the Missouri Ozarks, there has been a decrease in oaks from 79 percent of species composition to 57 percent of species composition. This has resulted in changes in communities. Dominant oak species (as defined by ≥ 10 percent composition) have been reduced and replaced by a mixture of many species, of which only eastern redcedar and maples have become dominant. Current forest densities are approximately two times greater than historical densities. Current forests are about 300 to 400 trees/ha (d.b.h. ≥ 12.7 cm), with little deviation. Historical forests had greater deviation and ranged from a mean of 75 to 320 trees per ha. The greatest density increases were in the drier western Ozarks close to the prairies, where historical densities were lower.

Although climate change, insects, fungal pathogens, and deer herbivory probably all contribute to oak declines, fire appears to be the primary factor that determines community composition and structure. Regionally distinct, oak-dominated and fire-dependent communities have been replaced by denser, fire-intolerant species. Currently, many oak-dominated forests have understories of shade-tolerant species, and in the future, upland eastern forests may become duplicates of mesic forests without management. Oak restoration will become more difficult if the overstory is no longer oak. Oak forest management and research priorities include commitment to both prescribed burns and opening the canopy and subcanopy where there is an oak understory.

LITERATURE CITED

ECOMAP. 1993. **National hierarchical framework of ecological units.** Washington, DC: U.S. Department of Agriculture, Forest Service. 20 p.

RESTORING OAK REGENERATION AFTER OVERGRAZING AND FIRE IN ZAGROS FORESTS

Azad Henareh Khalyani, Audrey L. Mayer, and Manouchehr Namiranian

Research Assistant (AHK) and Assistant Professor (ALM), School of Forest Resources and Environmental Science, Michigan Technological University, 1400 Townsend Dr., Houghton, MI, 49931; Professor (MN). AHK is corresponding author: to contact, call 906-231-1186 or email at ahenareh@mtu.edu.

INTRODUCTION

Zagros oak forests have a long history of use by local communities. Palynological research provides evidence of stock breeding, grazing, and agriculture at least since the beginning of the fifth millennium cal B.P. (Djamali and others 2009, Sumner 1990, Wright and others 1967). The traditional sustainable management was maintained by a homeorhetic regulation in pre-technological land use (Naveh 1988). This regulation was acquired by a balance between the anthropogenic utilization of the forests for livelihood necessities and stabilizing feedbacks from a lack of food, water, and social components, generating a resource protection mechanism. Resource utilization activities such as coppicing and cultivation were practiced by rotations (Ghazanfari and others 2004). Landscapes maintained a patchwork formed by the combined effects and interaction of fire, grazing, and coppicing.

However, contemporary technological progress and maladaptive policies have eliminated the economical feedbacks from population control by providing food, water, and medicine for man and livestock regardless of water supply, even though the lack of economical development should provide economic incentives to reduce pressures on the forests. The social stabilizing feedback was eliminated by a transition to governmental ownership and new regulations. Forests were nationalized in 1963, and grazing and coppicing became illegal. The Iranian government has tried to protect the forest (and encourage regeneration) from livestock grazing through the use of various exclosures, but these have not been effective in prohibiting overutilization in all areas because of the economical needs of locals. In this situation, the landscapes are no longer formed by the combined effects and interaction of fire, grazing, and coppicing because these influential factors are separated from each other in the landscape and have led to the formation of the two extreme models in Zagros landscapes: with and without exclosures. In areas without government exclosures, there is no or very limited fire frequency because of the quick consumption of understory grasslands by livestock. But overgrazing by livestock (mainly goats) leads to the lack of regeneration with consequent soil erosion. The additional pressure from climate change-induced droughts further catalyzes the transition of the forest landscapes to deserts. In areas with government exclosures that restrict access of people and livestock, the forest cover is frequently affected by fires during the dry season. Forest stands cannot recover quickly from the wildfires which devastate the ecological condition because of the dry and hot conditions in summer. If the fires are too frequent, they also reduce fauna and flora and inhibit regeneration through topsoil destruction.

The social and economical feedbacks, therefore, need to be restored in order to reactivate the dynamic self-stabilization and maintain ecological sustainability in patchy landscapes formed by the interaction of anthropogenic uses and fire. Restoration projects thus need to be carefully targeted, socially accepted, and implemented by direct participation of local communities using their traditional ecological wisdom. In this study we implement an adaptive management approach to: (1) gradually eliminate the boundaries between the two extreme conditions by reintroducing livestock grazing to avoid frequent extreme wildfires; (2) establish seed based regeneration in the context of grazing and coppicing by local communities; (3) restore the social feedbacks by giving access and utilization rights to local communities and use their participation in the project; and (4) evaluate the potential of the sites for regeneration establishment and compare different technical approaches for the regeneration establishment in the area.

STUDY AREA

The study area was 15 ha of Armardeh forests (45°48’ N, 35°56’ E) in the northern Zagros region in Baneh city, Kurdistan province, Iran (Fig. 1). These xeric forests are dominated by *Quercus* spp. with other deciduous species occurring occasionally. Forest cover rarely exceeds 70 percent. The average elevation is 1675 m a.s.l., and average annual precipitation is 615 mm. The bedrock is mainly formed by quartzite. The soil type is mainly formed of inceptisols and in some areas is heavily destructed and forms entisols.

METHODS

Thirty 1000 m^2 circular plots were randomly located in the study area. The height, diameter at breast height (d.b.h.), and crown diameter at two directions were measured for all trees in the circular plots (Fig. 1). One soil profile was dug in each plot, and soil physicochemical characteristics were analyzed. Four restoration techniques were established in each of

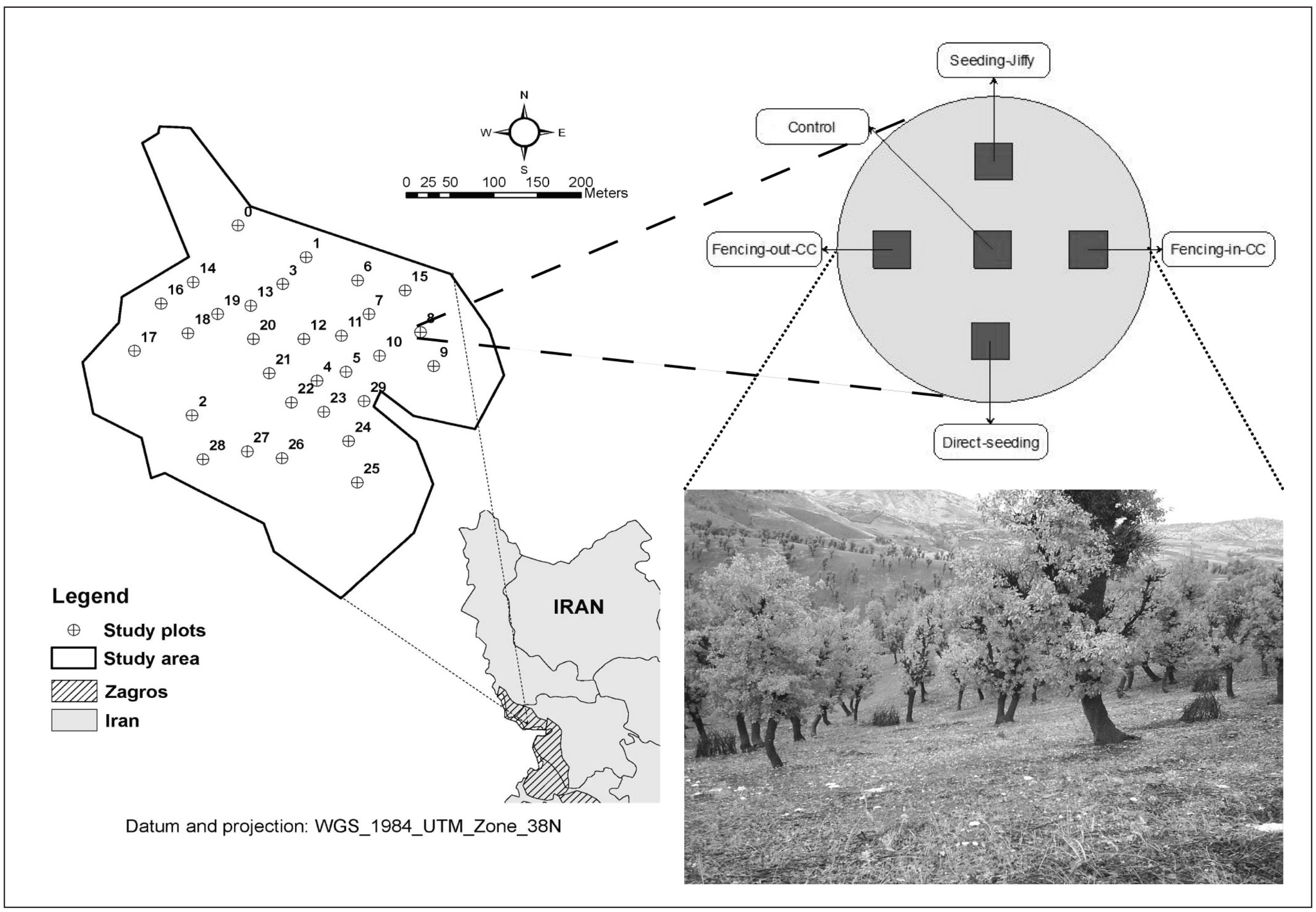

Figure 1.—Location of study area in Iran, position of the 1000 m^2 circular plots in the study area, and the five 1 m^2 micro plots in each circular plot.

the circle plots as study treatments in 1 m^2 micro plots, along with a control micro plot at the center of the plots in autumn. The study was conducted in the framework of a randomized complete block design (RCBD). The restoration approaches (treatment micro plots) were:

1. Fencing outside the crown coverage
2. Fencing under the crown coverage
3. Direct seeding outside the crown coverage
4. Seeding using Jiffy technique (Skoupy and Hughes 1971) outside the crown coverage

The treatment micro plots were fenced, and the control micro plots were left unfenced under the crown coverage of each plot's center tree (Fig. 1). The response variables were the proportion of established seedlings and the heights of the saplings in the spring. Analysis of Variance (ANOVA) and Tukey's post hoc test were used to test the difference between treatments at 0.05 significance level.

RESULTS

The quantitative measurements showed that 74.1 percent of the trees in the study area were Lebanon oak (*Quercus libani* Olivier). Gall oak (*Quercus infectoria* Olivier), Mann oak (*Quercus brantii* L.), pistachio (*Pistacia atlantica* Desf.), and wild pear (*Pyrus glabra* Boiss.) formed 24.8 percent, 0.6 percent, 0.3 percent, and 0.2 percent of the trees, respectively. Mean d.b.h., height of the trees, and canopy cover were 29 cm, 6.6 m, and 1314 m^2/ha, respectively. Aspects did not show significant effects on tree characteristics and treatment effects. Physicochemical analysis of soil characteristics in two layers showed that in spite of being mechanically compacted and destructed, it had proper physicochemical characteristics for regeneration (Henareh 2005, Namiranian and others 2007). There was a significant difference in restoration success between fenced and unfenced study plots (Fig. 2). Areas in exclosures demonstrated regeneration while controls with no fencing did not. Planting with Jiffy pellets resulted in significantly taller saplings compared to other approaches (Fig. 3). Direct seeding treatments were significantly lower in height than planting Jiffy pellets but taller than other treatments.

DISCUSSION

Higher percentage of Lebanon oak indicates forest stands with better ecological conditions, since this species requires higher humidity and deeper soils compared to other oak species. Forest stand measurements and soil analysis indicated that the potential for natural regeneration still exists. Applying exclosures over large areas increases the understory

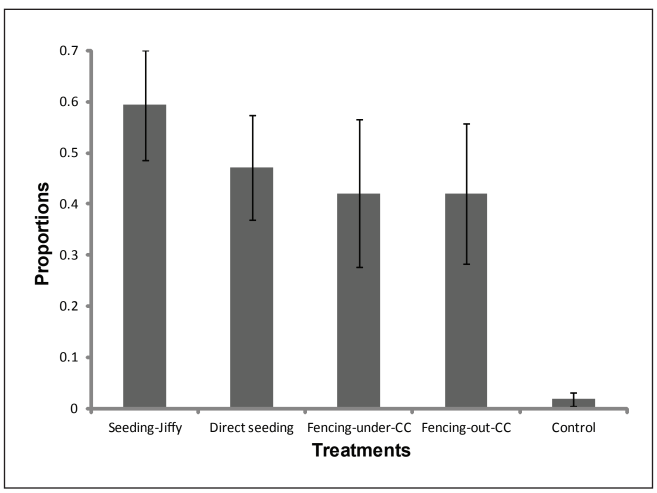

Figure 2.—Mean proportion of established seedlings for the five treatments in early spring (error bars represent standard errors).

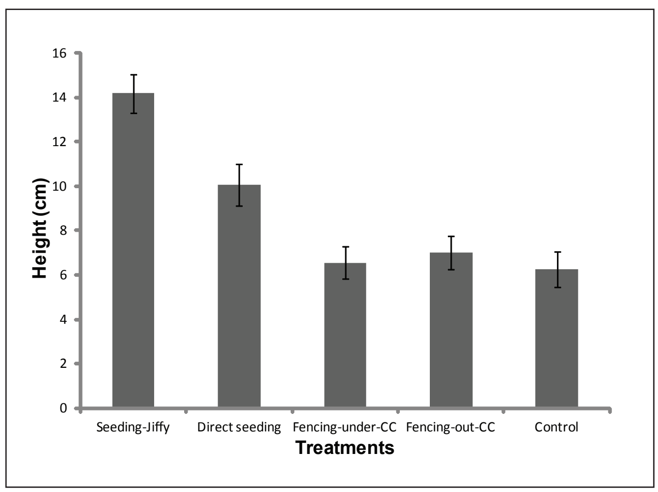

Figure 3.—Mean heights of saplings for the five treatments in summer (error bars represent standard errors).

dry vegetation and risk of extreme wildfires as well as social problems by forcing local communities out of forests. However, the very small exclosures tested here do not suffer from either of these problems and demonstrated that regeneration in a context of grazing and coppicing is feasible. Fencing at a small scale (1 m^2), even outside of the crown coverage and without applying artificial restoration techniques, ensured the establishment of regeneration significantly higher than in unfenced areas, where no regeneration occurred. No significant difference was found between natural and artificial restoration methods regarding the proportion of established seedlings. This suggests that the simple treatment of building a small fence around small, scattered areas is a low-cost, practical management approach that local communities can use to help restore forest regeneration into these landscapes. However, a significant difference was found between artificial and natural methods regarding the heights of the resultant saplings in the summer; the artificial methods resulted in taller samplings. These artificial methods may be a necessary addition in areas that have lost a significant amount of topsoil and where natural regeneration might grow too slow relative to the need for the resource base that the forests represent.

The study demonstrated that conservation at small scales in a context of livestock grazing and coppicing by local communities can result in forest regeneration and fire avoidance, while simultaneously being acceptable to local communities. Applied across the region, this approach could eliminate the boundary between the two dominant extreme conditions of fire and grazing and maintain patchy landscapes formed by the interaction of all factors including fire, grazing, and coppicing.

The content of this paper reflects the views of the author(s), who are responsible for the facts and accuracy of the information presented herein.

LITERATURE CITED

Djamali, M.; De Beaulieu, J.L.; Miller, N.F.; Andrieu-Ponel, V.; Ponel, P.; Lak, R.; Sadeddin, N.; Akhani, H.; Fazeli, H. 2009. **Vegetation history of the SE section of the Zagros Mountains during the last five millennia; a pollen record from the Maharlou Lake, Fars Province, Iran.** Vegetation History and Archaeobotany. 18(2): 123-136.

Ghazanfari, H.; Namiranian, M.; Sobhani, H.; Marvi Mohajer, R. 2004. **Traditional forest management and its application to encourage public participation for sustainable forest management in the northern Zagros Mountains of Kurdistan province, Iran.** Scandinavian Journal of Forest Research. 19(4): 65-71.

Henareh, A. 2005. **Investigation, monitoring and evaluation of forest sustainability using permanent plots and suggested applicable approaches in forest management in Zagros, case study in Armardeh.** Tehran, Iran: University of Tehran. 73 p. M.S. thesis. In Persian.

Namiranian, M.; Henareh, A; Ghazanfari, H.; Zahedi, G. 2007. **Investigation of various management approaches for restoration & regeneration settlement in northern Zagros.** Forest and Poplar Research. 15 (4): 386-397. In Persian.

Naveh, Z. 1988. **Biocybernetic perspectives of landscape ecology and management.** In: Moss M.R., ed. Landscape ecology and management. Montreal: Polyscience: 23-24.

Skoupy, J.; Hughes, E.L. 1971. **Reforestation using Jiffy 7 peat pellets.** Tree Planters' Notes. 22: 1-4.

Sumner, W.M. 1990. **An archaeological estimate of population trends since 6000 B.C. in the Kur River Basin, Fars Province, Iran.** In: South Asian archaeology. Rome, Italy: Istituto Italiano per il Medio ed Estremo Oriente: 1-16.

Wright, H.E.; McAndrews, J.H.; van Zeist, W. 1967. **Modern pollen rain in western Iran, and its relation to plant geography and Quaternary vegetational history.** Journal of Ecology. 55: 415-443.

INITIATING RESTORATION OF WOODLAND COMMUNITIES: THE EFFECT OF THINNING AND PRESCRIBED FIRE ON STAND STRUCTURE IN THE OZARK HIGHLANDS

Carter O. Kinkead, John M. Kabrick, Michael C. Stambaugh, and Keith W. Grabner

Graduate Research Assistant (COK); Research Forester (JMK), U.S. Forest Service, Northern Research Station, 202 Natural Resources Building, Columbia, MO 65211; Research Associate (MCS); Ecologist (KWG). JMK is corresponding author: to contact, call 573-875-5341 ext. 229 or email at jkabrick@fs.fed.us.

INTRODUCTION

In the last two centuries, Missouri Ozark forests have undergone immense transformation as a result of timber harvests and fire suppression (Dey and Hartman 2005, Ladd 1991). As fuel accumulation and canopy closure have increased (Kolaks 2004), so has interest in restoring and maintaining the woodland and savanna ecosystems that once prevailed across the Ozark landscape (Johnson and others 2002, Ladd 1991, Nuzzo 1986). Woodland communities are characterized by open understories and dense ground flora comprised of forbs, grasses, and sedges (Nelson 2005, Nuzzo 1986, Taft 2009). They once were common in the western Central Hardwood Region and prairie-forest transition zone where low-intensity fires occurred frequently (Guyette and others 2002, Taft 2009). With fire-based management making its way to the forefront of this effort, a number of science-based efforts are aimed at quantifying the response of natural systems to various fire or disturbance regimes.

The purpose of this study was to gain insight as to the implications of various treatments, from prescribed burning to thinning harvests and the combination. Our objective was to examine changes in forest structure, such as diameter distribution, canopy cover, and species composition, as a function of fire, thinning, and aspect to better understand how to restore oak and oak-pine woodlands.

The project area is located in the Missouri Ozarks at Logan Creek and Clearwater Creek Conservation Areas, both managed by the Missouri Department of Conservation. Each site lies within the Black River Oak/Pine Woodland/Forest Hills Ecological Landtype Association that is characterized by steep hillslopes with cherty, low-base soils (Nigh and Schroeder 2002). The fully stocked, second-growth forests on each site have had no known management or fire for over 40 years.

METHODS

Four treatments (burn, harvest, harvest-burn, and control) were applied to three (north slope, ridge, and south slope) ecological landtypes (ELTs) across three replicated blocks. Treatments were paired with the ELT to create 12 experimental units per block. Timber harvests (overstory thinning) occurred during the summer and early fall 2002 prior to the first burn. Harvests reduced stocking to approximately 45 ft^2/acre by thinning from below, leaving primarily the dominant trees with preference for retaining shortleaf pine (*Pinus echinata* Mill.) and white oak (*Quercus alba* L.). We use the term “thinning” because the goal was not to regenerate the stand but rather to enhance the growth of the residual stand including the forbs, legumes, and graminoids in the understory. Two prescribed fires have been conducted on burn and harvest-burn units in each of the three blocks. The first occurred in early April of 2003 and the second in April 2005 for blocks one and two. Because of unsuitable weather conditions, block three was not burned until March 2006. Consequently it was excluded from analysis.

Pretreatment data were collected in the summer of 2001, and posttreatment data were collected in the summer of 2003 and 2005. Permanent vegetation plots were installed along transects during the summer of 2001. Overstory vegetation plots consisted of three 1/3 acre circular plots randomly located within each treatment/aspect class. All live trees (diameter at breast height [d.b.h.] > 1.5 inch) were identified to species and d.b.h. was measured. Ground flora data were collected at a total of 1,080 quadrats (size = 10.8 ft^2, 30 per experimental unit). Within each quadrat, all live herbaceous plants and shrubs were identified to species, and cover by species was estimated to the nearest percent.

RESULTS AND DISCUSSION

Average pretreatment stocking for all stands was 94.2 percent, composed primarily of trees greater than 10.5 inches d.b.h., even after treatments. Prescribed fire alone decreased overall stocking nearly 10 percent from pretreatment conditions, having significant reductions in all trees less than 10 inches. In fact, 2-5 inch stems declined 22 percent after a single fire, and 52 percent after a second fire. However, due to the minimal impacts on larger diameter trees, the effects of prescribed fire to overall stocking were not significantly different from the control. No significant ($P < 0.05$) interaction between aspect and treatment on stocking or ground flora coverage was detected.

The effects of harvest-burn treatments on stocking did not significantly differ from harvest only treatments (Fig. 1). These findings indicate that prescribed fire has the ability to decrease the abundance of small diameter stems without drastically altering stand structure. From a woodland restoration standpoint, it is desirable to decrease the abundance of understory woody plants to provide more growing space for the herbaceous plants. In the absence of fire, hardwood seedlings and seedling sprouts proliferate, which can shade herbaceous plants and reduce their coverage (Hutchinson and others 2005). This is especially the case in the presence of increased canopy openness, such as after a harvest. Although our results show a minor decrease in woody stems one year after harvest-burn treatments, preliminary results from recently collected data (2011) suggests that without continued burning, small trees and shrubs begin to limit all other life forms in the understory.

Life forms documented in the quadrats include forbs, graminoids, legumes, shrubs, and vines. All of these physiognomic groups increased with both prescribed fire and the combination of burning and harvesting. Since many restoration efforts are aimed at increasing forb and graminoid communities, it is important to note that harvest-burn treatments more than doubled the percent coverage of these life forms compared

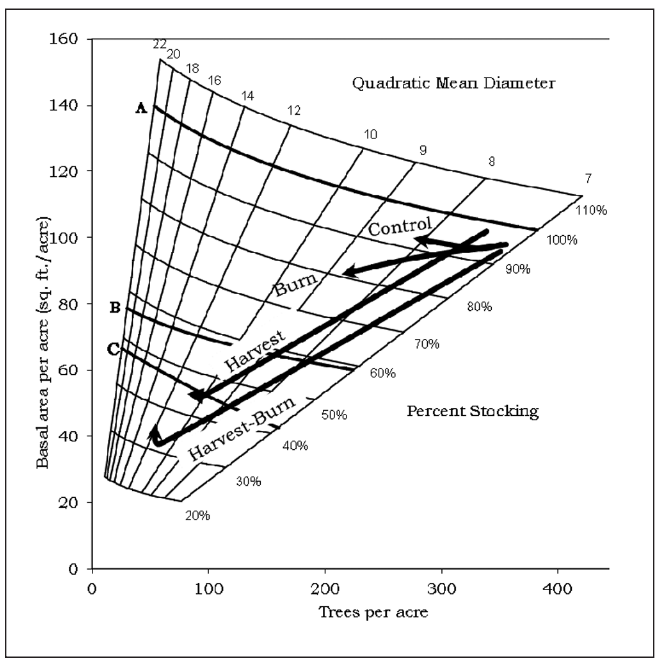

Figure 1.—Gingrich (1967) diagram portraying changes in stocking for each treatment from 2001 to 2006. Actual stocking percentages are slightly misrepresented because the parameters used are based on “typical upland oak-hickory” stands rather than calibrated to the quadratic mean diameter of our study site (Johnson and others 2002).

to burning alone and harvesting alone. Analyses of a class of woodland indicator species (i.e., a collective group of forbs, graminoids, legumes, and shrubs) (Table 1) showed this class increased 42 percent in relative cover on plots that were harvested and burned, a near three- to fourfold increase compared to burn only and harvest only treatments (Fig. 2). Woodland indicators within control plots increased 2 percent.

Overall, the results of this study suggest that commercial overstory thinning followed by two prescribed fires has the potential to restore woodland structure and composition (Nelson 2005, Taft 2009). Our findings also demonstrate the relationships between stand structural metrics such as stocking to woodland ground flora and shrub cover in the presence or absence of prescribed fire. These relationships have seldom been quantified and documented in the woodland restoration literature. This sequence of management techniques most effectively resulted in stands characteristic of woodland conditions (i.e., basal area 30-50 ft^2/acre, abundant forb and grass populations, and minimal midstory tree density [Nelson 2005]). While mechanical removal of the midstory and overstory trees fulfills these restoration objectives, increased burning becomes necessary to manage the understory woody response to canopy openness. Prescribed fire alone may pose a sustainable option for woodland restoration where overstory stocking is decreased over time, thereby subtly achieving canopy openness and suppression of woody regeneration.

Table 1.—List of woodland indicator species used in understory vegetation sampling

Andropogon gerardii
Asclepias tuberosa
Aureolaria grandiflora
Babtisia bracteata
Blephilia ciliata
Ceanothus americanus
Comandra umbellata
Coreopsis palmata
Cunila origanoides
Dalea purpurea
Desmodium rotundifolium
Echinacea pallida
Eryngium yuccifolium
Euphorbia corollata
Gentiana alba
Gillenia stipulata
Helianthus hirsutus
Ionactis linariifolius
Lespedeza hirta
Lespedeza procumbens
Lespedeza violacea
Lespedeza virginica
Liatris aspera
Liatris squarrosa
Lithospermum canescens
Monarda bradburiana
Orbexilum pedunculatum
Parthenium integrifolium
Phlox pilosa
Pycnanthemum tenuifolium
Schizachyrium scoparium
Silene regia
Silene stellata
Silphium integrifolium
Silphium terebinthinaceum
Solidago hispida
Solidago petiolaris
Solidago radula
Solidago rigida
Solidago speciosa
Solidago ulmifolia
Sorghastrum nutans
Symphyotrichum anomalum
Symphyotrichum oolentangiense
Symphyotrichum patens
Symphyotrichum turbinellum
Taenidia integerrima
Tephrosia virginiana
Verbesina helianthoides
Violoa pedata

LITERATURE CITED

Dey, D.C.; Hartman, G. 2005. **Returning fire to Ozark Highland forest ecosystems: effects on advance reproduction.** Forest Ecology and Management. 217: 37-53.

Gingrich, S.F. 1967. **Measuring and evaluating stocking and stand density in upland hardwood forests in the Central States.** Forest Science. 13: 38-53.

Guyette, R.P.; Muzika, R.M.; Dey, D.C. 2002. **Dynamics of an anthropogenic fire regime.** Ecosystems. 5: 472-486.

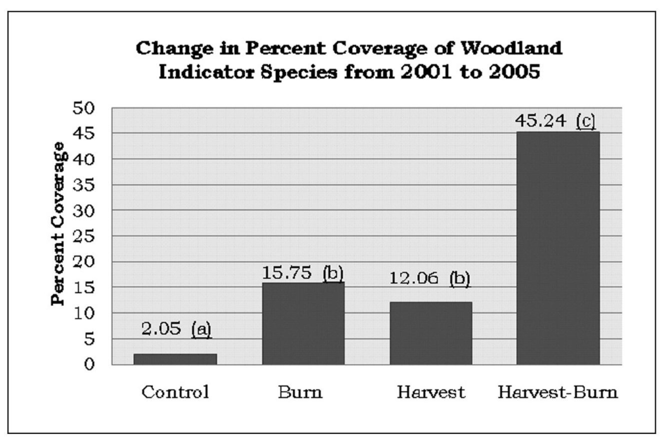

Figure 2.—Changes in percent coverage of woodland indicator species (see Table 1) for all treatments. Significant differences (P < 0.05) among treatments are indicated with differing letters.

Hutchinson, T.F.; Sutherland, E.K.; Yaussy, D.A. 2005. **Effects of repeated prescribed fires on the structure, composition, and regeneration of mixed oak forests in Ohio.** Forest Ecology and Management. 218: 210-228.

Johnson, P.S.; Shifley, S.R.; Rogers, R. 2002. **The ecology and silviculture of oaks.** Wallingford, UK: CABI. 503 p.

Kolaks, J. 2004. **Fuel loading and fire behavior in the Missouri Ozarks of the Central Hardwood Region.** Columbia, MO: University of Missouri. 115 p. M.S. thesis.

Ladd, D. 1991. **Reexamination of the role of fire in Missouri oak woodlands.** In: Burger, G.V.; Ebinger, J.E.; Wilhelm, G.S., eds. Proceedings of the oak woods management workshop; 1988 October 21-22; Charleston, IL: 67-80.

Nigh, T.A.; Schroeder, W.A. 2002. **Atlas of Missouri ecoregions.** Jefferson City, MO: Missouri Department of Conservation. 212 p.

Nelson, P.W., 2005. **The terrestrial natural communities of Missouri.** Jefferson City, MO: Missouri Natural Areas Committee. 550 p.

Nuzzo, V.A. 1986. **Extent and status of Midwest oak savanna: pre-settlement and 1985.** Natural Areas Journal. 6: 6-36.

Taft, J.B. 2009. **Effects of overstory stand density and fire on ground layer vegetation in oak woodland and savanna habitats.** In: Hutchinson, T.F., ed. Proceedings of the 3rd fire in eastern oak forests conference; 2008 May 20-22; Carbondale, IL. Gen. Tech. Rep. NRS-P-46. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station: 21-39.

EFFECTS OF MULTIPLE INTENSE DISTURBANCES AT MANLEY WOODS, WILSON'S CREEK NATIONAL BATTLEFIELD

Sherry A. Leis and Kevin James

Fire Ecologist (SAL), Biology Department, Missouri State University, 901 South National Avenue, Springfield, MO 65897; Plant Ecologist (KJ). SAL is corresponding author: to contact, call 417-836-8919 or email at sleis@partner.nps.gov.

ABSTRACT

Natural communities in southwest Missouri are often challenged by ecological disturbances of a catastrophic nature. Windthrow, wildfire, flooding, drought, and ice storm events all occur periodically and range in spatial extent from local to regional. Although in this region local disturbances are common in singularity, it is unusual for several disturbances to affect the same locality within a short time period. Since 2003, however, the Ozark Highlands region has seen widespread events including tornados, fires, floods, and ice storms, many occurring in succession on the same piece of land. Understanding the change in vegetation composition and recovery rates resulting from multiple severe disturbances is crucial to natural resource management planning.

In 2003, a tornado struck the Manley Woods area of Wilson's Creek National Battlefield (WICR) affecting 60 ha. The affected area included long-term monitoring sites sampled both 3 years prior to and post-tornado. In the post-tornado years, three additional notable disturbances occurred, but each disturbance differentially affected each of the sites. Increased fuel loads resulting from the tornado damage caused concern for elevated wildfire risk, so park staff made plans to reduce fuel loads. In 2005, a contractor removed salvageable timber from Manley Woods, and the following year a prescribed fire was completed. In 2007 a January ice storm further affected communities in Manley Woods. The aim of this study was to determine the effectiveness of the fuels reduction activities, and to describe the composition of overstory and understory plant communities currently inhabiting Manley Woods.

Fuel loads at Manley Woods were quite high post-tornado, especially at site four (Fig. 1). The fuels reduction process, log removal, and fire reduced mean fuel loads by approximately half, leaving an average of 18.5 tons/ha. Mean fuel loads were significantly reduced at site 4 that received the most damage and notable reductions were seen at site 5.

Changes occurred in several aspects of the plant community. Overstory tree density was reduced by 4.31 m^2 per ha, mainly by the tornado (Fig. 2). In 2006, trees in the larger size classes (35+ cm diameter at breast height [d.b.h.]) could only be found at the less disturbed sites in the woods, and canopy closure trended towards a savannah structure. Regeneration (seedlings and saplings) was reduced overall, including a significant decline in oak and hickory species. Diversity of the herbaceous understory plants dropped significantly during the period of heaviest disturbance (2003-2006). Understory plant community response lagged 1 year behind disturbance events (Fig. 3). Plant

community composition shifted towards more early seral woodland species after 2004 at all sites, but the greatest change occurred at the most heavily disturbed areas in the woods. Furthermore, 2007 composition appeared to be on a path back toward previous community composition. Plant guilds, particularly woody plants, pulsed after the tornado but returned to similar predisturbance conditions. Exotic species remained low and did not change significantly over time.

This analysis may aid park natural resource managers to plan to achieve the desired conditions in Manley Woods. Prior to the 2003 tornado, park managers used fire to work towards a savannah community type. The disturbances had a heterogeneous effect on the whole management unit, hastening the restoration process in parts of Manley Woods, but not greatly affecting others.

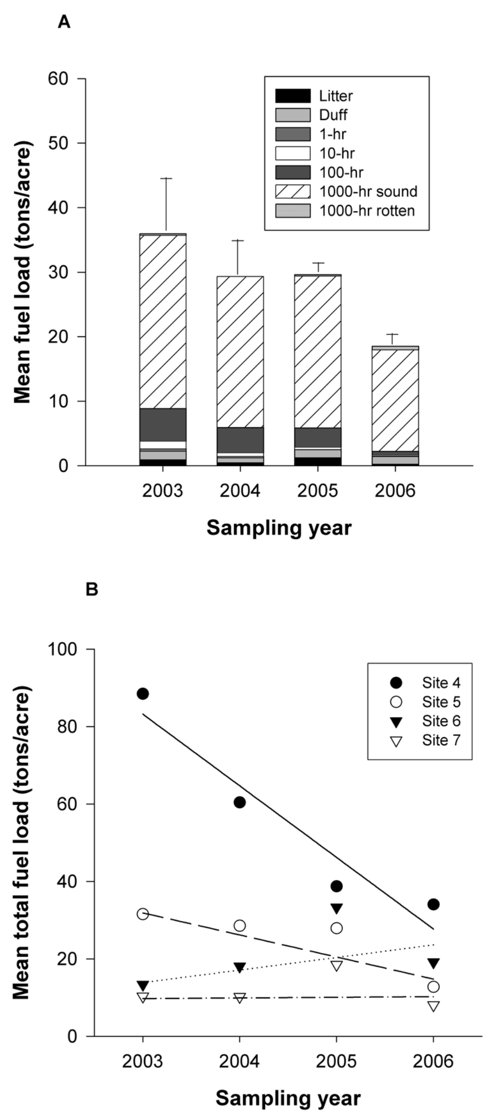

Figure 1.—(A) Mean fuel loading by time lag class for Manley Woods. Error bars represent one standard deviation for mean total fuel load (not time lag class), (B) mean fuel loading by sampling site for Manley Woods. Asterisk indicates loadings at site 4 were significantly different through time ($P = 0.04$, $r^2 = 0.92$).

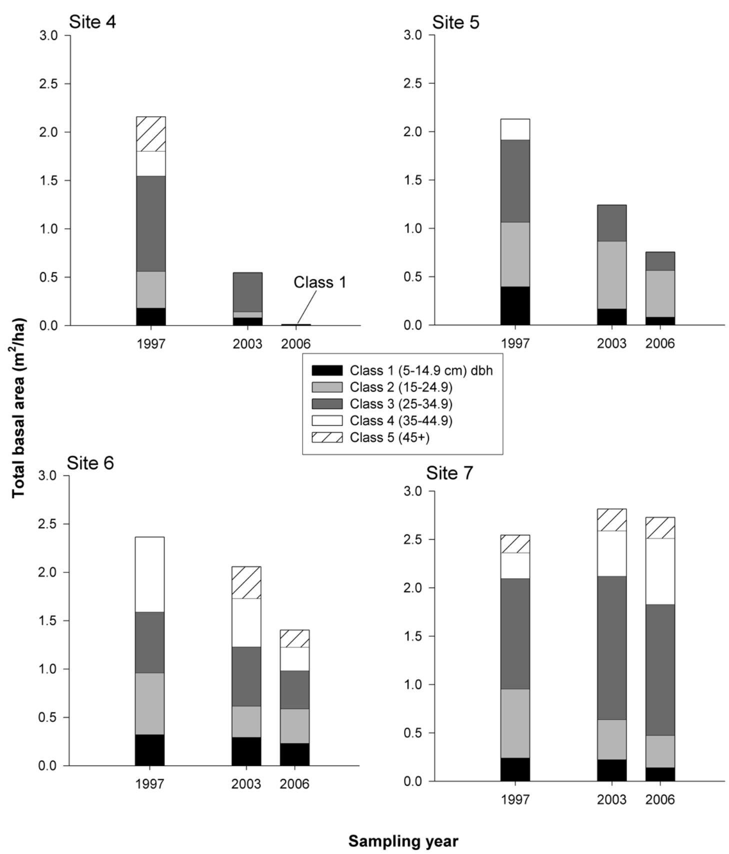

Figure 2.—Total basal area (m²/ha) of trees in each size class throughout monitoring history at Manley Woods, (diameter at breast height = d.b.h.).

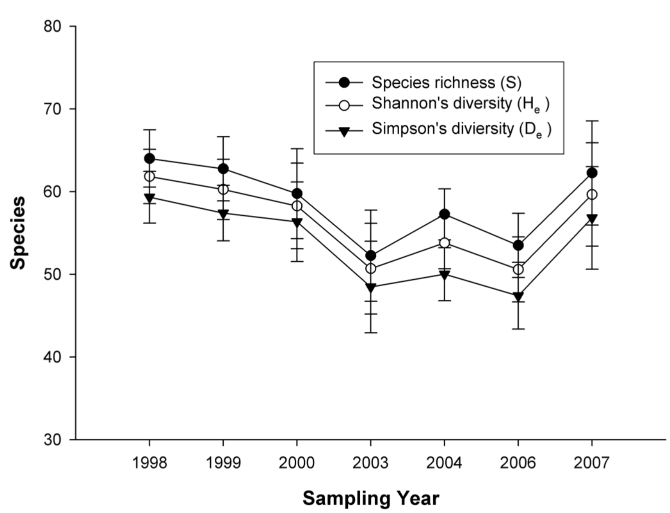

Figure 3.—Mean diversity measures with standard error bars for the understory community in Manley Woods long-term monitoring plots. Measures were converted to effective number of species for comparison.

POTENTIAL EFFECTS OF FIRE AND TIMBER HARVEST ON TERRESTRIAL SALAMANDER MICROHABITAT USE IN A MISSOURI OZARK FOREST

Katherine M. O'Donnell, Frank R. Thompson III, and Raymond D. Semlitsch

Graduate Student Researcher (KMO), Division of Biological Sciences, University of Missouri, 105 Tucker Hall, Columbia, MO 65211-7400; Research Wildlife Biologist (FRT); Curators Professor (RDS). KMO is corresponding author: to contact, call 573-882-1421 or email at kmozrc@mail.missouri.edu.

BACKGROUND

Prescribed fire and timber harvest are human-caused disturbances that can have substantial effects on forest ecosystems. It is generally accepted that some methods of timber harvest adversely affect forest-dependent wildlife, including amphibians (e.g., Petranka and others 1993, Semlitsch and others 2009). However, wildlife responses to fire are not as well understood; they vary considerably among species, and are affected by fire seasonality and severity (Lyon and others 2000). As the use of prescribed fire becomes more common, it is essential to better understand the effects of controlled burns on forest-dependent wildlife.

Fire and Amphibians

Fire effects on amphibians remain incompletely understood but likely vary among species, geographic regions, and habitat types (Pilliod and others 2003). Amphibians require wet or moist places for breeding, nesting, and respiration, and are very susceptible to heat and desiccation. Therefore, they often retreat under cover objects such as rocks and logs to retain moisture when the leaf litter is dry. Amphibians have limited movement capacity (Russell and others 1999), which likely forces them to stay and cope with habitat changes postfire. This unique suite of characteristics may make amphibians more vulnerable to fire than other wildlife taxa.

Recent studies have found little evidence of amphibians being directly affected by prescribed fire (burns, mortality from fire) in the short-term (Pilliod and others 2003, Russell and others 1999). However, fire can indirectly affect wildlife by altering components of their habitat. These indirect effects are likely key for amphibians because of their sensitivity to changes in microhabitat, such as leaf litter and downed wood, and could be long-lasting (Pilliod and others 2003).

Focal Species

We chose to focus our study on the southern redback salamander (*Plethodon serratus*) (Fig. 1). Redback salamanders and other species in the Plethodontidae family are fully terrestrial; they do not have an

Figure 1.—Southern redback salamander (*Plethodon serratus*).

aquatic larval stage like many other amphibians. They undergo direct development resulting in hatchlings that resemble miniature adults. Plethodontid salamanders are also lungless and respire directly through their skin, and therefore, are very moisture-dependent. These salamanders typically have very small home ranges. Previous studies have shown that the southern redbacks are most surface-active in the spring and fall when temperatures are moderate (Herbeck and Semlitsch 2000). They spend most of the summer months in underground refugia to avoid desiccation and heat stress. They are also nocturnal and come to the surface at night to forage. During the day, they usually retreat underground or under cover objects.

Importance

Terrestrial salamanders like the southern redback are the dominant vertebrate predators in many forest ecosystems and represent a large portion of the vertebrate biomass (Burton and Likens 1975), although they are often overlooked because of their fossorial nature. Several studies suggest that these types of salamanders may play an integral role in forest ecosystem dynamics, such as nutrient cycling, leaf litter decomposition, and even carbon sequestration (Burton and Likens 1975, Welsh and Droege 2001, Wyman 1998). Disturbance events such as timber harvest cause drying of the soil and leaf litter and make forests less capable of supporting amphibians (Semlitsch and others 2009, Welsh and Droege 2001). Because prescribed fires also affect leaf litter and cover objects, there may be negative consequences for amphibians. We are investigating the effects of prescribed fire and timber harvest on terrestrial salamanders. We are interested in how these forestry practices affect salamander abundance and behavior. This paper focuses on the predicted outcomes of the study, based on pretreatment data, on the distribution of salamanders among microhabitats.

STUDY AREA AND METHODS

The study is taking place in the Sinkin Experimental Forest, located within the Mark Twain National Forest in the Ozark Highlands region of south-central Missouri. The site consists of mature, fully-stocked oak-hickory stands. Five 5-ha replicate experimental plots per treatment will be subjected to one of the following treatments: prescription burn, shelterwood harvest, or midstory herbicide. Five plots will serve as controls. We sampled two locations (upslope, downslope) within each of the 20 sampling plots in the spring and fall of 2010. We conducted 3 x 3 m area-constrained searches of natural cover objects and leaf litter, and measured snout-vent length (SVL) and recorded the capture location of all salamanders (southern redback salamander, western slimy salamander [*Plethodon albagula*], and dark-sided salamander [*Eurycea longicauda melanopleura*]).

RESULTS

We captured 1025 southern redback salamanders, 18 western slimy salamanders, and 4 dark-sided salamanders. We found most salamanders (74 percent) within the leaf litter; the remaining 25 percent were found under natural cover objects such as logs and rocks (Fig. 2). We captured more salamanders during downslope surveys (566) than upslope surveys (481). Salamander density was strongly correlated with recent rainfall (Fig. 3). We also found no significant pretreatment differences between groups.

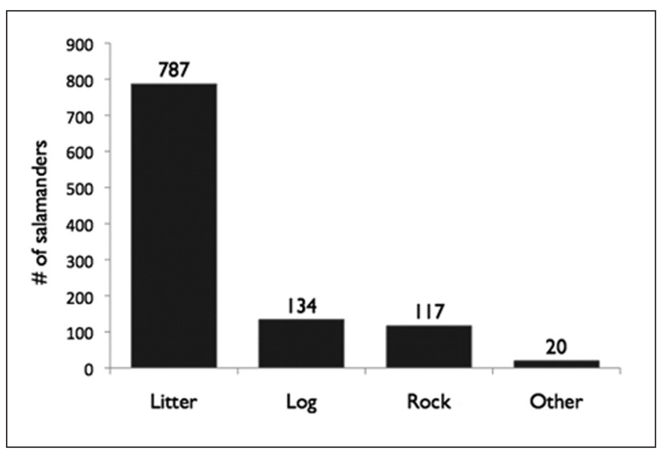

Figure 2.—Location of salamander captures.

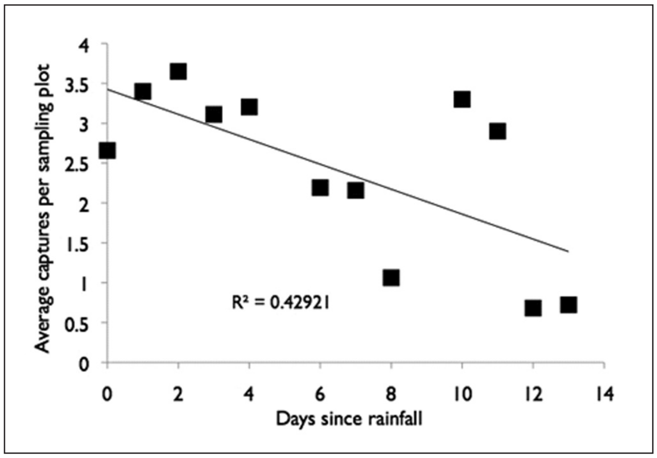

Figure 3.—Relationship between number of salamander captures and recent rainfall.

CONCLUSIONS AND PREDICTIONS

Our pretreatment results suggest that redback salamanders are more likely to be found in leaf litter than expected. Many studies on terrestrial salamanders focus on sampling natural cover objects and do not include leaf litter searches. We believe our sampling technique results in a more complete understanding of salamanders' distribution among microhabitats, and it will allow us to detect changes in cover object use following prescribed fire and timber harvest. We predict that competition for cover objects will increase following fire and timber harvest due to decreases in leaf litter and drier surface conditions. We expect to reveal more about the role of leaf litter loss in salamander population persistence.

It is important for managers to consider the unique characteristics of amphibians when implementing forest management plans. Because amphibians have different microhabitat requirements than other vertebrate taxa such as lizards or snakes, they are likely to have different responses to fire. Terrestrial salamanders, such as the southern redback, may be even more susceptible to fire than other amphibians.

ACKNOWLEDGMENTS

This project is funded by a grant from the U.S. Department of Agriculture, Forest Service (#10-JV-11242311-061). Many thanks to Dana Drake, Michael Osbourn, Grant Connette, and Katie LaJeunesse for assistance in the field. Photo provided by Bill Peterman.

LITERATURE CITED

Burton, T.M.; Likens, G.E. 1975. **Salamander populations and biomass in the Hubbard Brook Experimental Forest, New Hampshire.** Copeia. 1975(3): 541-546.

Herbeck, L.A.; Semlitsch, R.D. 2000. **Life history and ecology of the southern redback salamander, *Plethodon serratus*, in Missouri.** Journal of Herpetology. 34(3): 341-347.

Lyon, L.J.; Telfer, E.S.; Schreiner, D.S. 2000. **Direct effects of fire and animal responses.** In: Smith, J.K., ed. Wildland fire in ecosystems: effects of fire on fauna. Gen. Tech. Rep. RMRS-42-vol. 1. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 17-23.

Petranka, J.W.; Eldridge, M.E.; Haley, K.E. 1993. **Effects of timber harvesting on Southern Appalachian salamanders.** Conservation Biology. 7(2): 363-370.

Pilliod, D.S.; Bury, R.B.; Hyde, E.J.; Pearl, C.A.; Corn, P.S. 2003. **Fire and amphibians in North America.** Forest Ecology and Management. 178: 163-181.

Russell, K.R.; Van Lear, D.H.; Guynn, D.C. 1999. **Prescribed fire effects on herpetofauna: review and management implications.** Wildlife Society Bulletin. 27(2): 374-384.

Semlitsch, R.D.; Todd, B.D.; Blomquist, S.M.; Calhoun, A.J.K.; Gibbons, J.W.; Gibbs, J.P.; Graeter, G.J.; Harper, E.B.; Hocking, D.J.; Hunter, M.L.; Patrick, D.A.; Rittenhouse, T.A.G.; Rothermel, B.B. 2009. **Effects of timber harvest on amphibian populations: understanding mechanisms from forest experiments.** BioScience. 59: 853-862.

Welsh, H.H.; Droege, S. 2001. **A case for using plethodontid salamanders for monitoring biodiversity and ecosystem integrity of North American forests.** Conservation Biology. 15(3): 558-569.

Wyman, R.L. 1998. **Experimental assessment of salamanders as predators of detrital food webs: effects on invertebrates, decomposition and the carbon cycle.** Biodiversity and Conservation. 7: 641-650.

SOIL AND LITTER NUTRIENTS FOLLOWING REPEATED AND PERIODIC BURNING IN THE MISSOURI OZARKS

Felix Ponder, Jr.

For more information, contact Daniel C. Dey, Research Forester, Northern Research Station, 202 ABNR Bldg., University of Missouri-Columbia, Columbia, MO 65211-7260. To contact, call 573-875-5341 or email ddey@fs.fed.us

ABSTRACT

Extractable nutrients in the surface litter and soil of oak-hickory (*Quercus* spp. and *Carya* spp.) forest plots subjected to annual and periodic prescribed burning were studied on plots located in the Chilton Creek basin in southeast Missouri on land owned and managed by The Nature Conservancy (TNC). Soils are generally poor and excessively drained and have developed over dolomite and sandstone bedrock. The TNC has intensively monitored vegetation responses to the high frequency, low intensity fire regime at the site including such variables as frequency, timing, intensity, and fuel conditions since pretreatment in 1996 and posttreatment in 1998 (Table 1). Little work has been performed at the Chilton Creek Management Area to examine soil and litter nutrient changes that may result from fire activities on the highly weathered soils on the site.

In 2007, the crew monitoring vegetation was permitted to collect soil and litter samples on vegetation plots during the same visit. After processing samples, they were analyzed for various chemical properties including pH, total organic carbon (TOC), N, P, K, Ca, Mg, and S.

Nitrogen and S in the upper 0 to 15 cm of soil were significantly lower ($\alpha = 0.05$) for plots burned annually than in plots burned periodically (Table 2). Several nutrients (P, K, Ca, Mg, and S) in the surface litter also differed between burning treatments. Differences were not consistent among treatments, suggesting possible differences in plant species contributing to litter composition as a consequence of burning treatments.

Differences in nutrients attributed to aspect showed litter N, P, and K and soil N, P, K, and S were lowest on north facing slopes and usually highest on south facing slopes (Table 3). Differences in site moisture and temperature between aspects are likely explanations for the response. Correlations for aspect were significant for soil P ($r = 0.45$, $P = 0.016$) and for litter S ($r = 0.45$, $P = 0.034$). Except for N and P, concentrations of nutrients in litter were not strongly related to the concentration of nutrients in the soil. Further, among nutrients, only the concentration of P in the soil and its corresponding level in the litter were significantly correlated ($r = 0.49$, $P = 0.012$).

Overall, TOC, although somewhat lower in soil for the annual burn treatment, did not differ between burn treatments for soil or litter. Unfortunately, this study does not include a “control” or “no burn” treatment for comparison. However, considering the lack of a control treatment, based on soil nutrient concentrations and litter nutrition, these results suggest that annual and periodic burning has had minimal effects on soils within the Chilton Basin, and perhaps the most ecologically significant effect was the loss of soil

N and S associated with annual burning. While these data represent a one-time view after 9 years of annual and periodic prescribed burns, they do give insight on soil and litter nutrient changes associated with burning treatments within the Chilton Creek Management Area in the Missouri Ozarks.

ACKNOWLEDGMENTS

The author thanks the Nature Conservancy for permission to do this study.

Table 1.—Burn schedule for fire units in the Chilton Preserve (Missouri Ozarks)

Location	Aspect	Size (ac)	Treatment (Burn interval over 9 years) 1	2	3	4	5	6	7	8	9	Sequence	Return Interval
Kelly	North	600	X	X	X	X	X	X	X	X	X	Annual	1.00
Kelly	South	410	X		X	X			X			1.3.4.7	2.25
Chilton	East	460	X				X	X			X	1.5.6.9	2.25
Chilton	North	465	X	X	X			X				1.2.3.6	2.25

Table 2.—Effect of fire sequence and interval on soil quality characteristics in the upper soil layer in the Chilton Creek Preserve located in the Missouri Ozarks. EC=electrical conductivity. Means plus standard errors in parentheses within a row followed by different letters are significantly different at α = 0.05 (Tukey's HSD).

	Burn sequence Interval (Years burned over 9 years)				
Property	Annually 1.00	1.3.4.7 2.25	1.5.6.9 2.25	1.5.7 3.00	1.2.3.6 2.25
Soil					
pH	5.3(1.0)	5.8(0.9)	5.5(0.8)	4.9(0.4)	5.0(0.9)
EC	132.4(112)	117(66)	120.6(21)	83.4(14)	77(41)
Carbon (g kg^{-1})	1.5(0.5)	2.7(0.9)	1.8(1.5)	2.6(0.9)	2.3(0.7)
N (%)	0.08(0.03)b	0.17(0.08)a	0.11(0.03)ab	0.10(0.02)ab	0.12(0.02)ab
P (mg kg^{-1})	10.4(4.4)	17.8(4.1)	15.9(3.3)	12.2(3.5)	13.7(6.8)
K (mg kg^{-1})	0.6(10)	85.0(43)	85.7(14)	47.1(15)	61.0(46)
Ca (mg kg^{-1})	419(370)	833(621)	525(484)	184(129)	140(109)
Mg (mg kg^{-1})	106.5(74)	226.2(168)	69.9(55)	24.3(11)	36.1(18)
S (mg kg^{-1})	10.4(3.0)a	14.2(3.7)ab	15.1(4.5)ab	15.5(1.2)ab	17.4(4.6)b
Litter					
Carbon (g kg^{-1})	44.1(5.1)	43.8(2.3)	46.0(1.2)	46.0(2.1)	44.1(5.3)
N (g kg^{-1})	0.91(0.2)	1.2(0.1)	1.1(0.2)	0.99((0.1)	1.22(0.3)
P (g kg^{-1})	0.05(0.01)c	0.08(0.00)a	0.06(0.02)abc	0.05(0.00)bc	0.07(0.01)ab
K (g kg^{-1})	0.14(0.04)b	0.13(0.02)b	0.16(0.02)b	0.16(0.01)b	0.25(0.06)a
Ca (g kg^{-1})	1.5(0.4)ab	2.1(0.3)a	1.6(0.3)ab	1.2(0.5)b	1.3(0.4)ab
Mg (g kg^{-1})	0.20(0.07)a	0.14(0.05)ab	0.16(0.03)ab	0.10(0.03)b	0.14(0.03)ab
S (g kg^{-1})	0.09(0.02)ab	0.12(0.01)a	0.11(0.01)ab	0.08(0.01)b	0.10(0.02)ab

Table 3.—Influence of site aspect on soil quality characteristics in the upper soil layer (0 to 15 cm) and surface litter following burnings in the Chilton Creek Preserve located in the Missouri Ozarks. Means plus standard errors in parentheses within a row followed by different letters are significantly different at α = 0.05 (Tukey's HSD).

	Site aspect		
Property	**North**	**South**	**East**
Soil			
pH	5.2(0.8)	5.4(0.6)	5.5(0.9)
Ec	116.1(85)	98.5(43)	120.6(59)
Carbon (g kg-1)	1.9(1.4)	2.5(1.3)	1.8(0.9)
N (%)	0.09(0.04)b	0.14(0.07)a	0.11(0.6)ab
P (mg kg-1)	10.9(4.0)b	15.9(6.4)a	15.9(7.5)a
K (mg kg-1)	42.8(12.2)b	73.0(9.8)a	85.7(47.6)a
Ca (mg kg-1)	341.1(262)	513.2(354)	526.4(432)
Mg (mg kg-1)	79.1(42.5)	138.4(185.5)	69.9(57.1)
S (mg kg-1)	11.1(2.3)b	15.7(4.2)a	15.1(4.9)a
Litter			
Carbon (g kg-1)	44.7(3.2)	44.0(3.8)	46.0(1.2)
N (g kg-1)	0.93(0.12)b	1.21(0.017)a	1.12(0.16)ab
P (g kg-1)	0.05(0.01)b	0.07(0.01)a	0.06(0.02)a
K (g kg-1)	0.14(0.02)b	0.16(0.04)ab	0.20(0.02)a
Ca (g kg-1)	1.41(0.45)	1.63(0.33)	1.59(0.30)
Mg (g kg-1)	0.16(0.05)	0.14(0.04)	0.16(0.03)
S (g kg-1)	0.09(0.01)	0.10(0.01)	0.11(0.01)

HERBACEOUS UNDERSTORY AND SEED BANK RESPONSE TO RESTORATION EFFORTS AT AN OAK-PINE BARRENS IN CENTRAL WISCONSIN

Jeffrey L. Ralston and James E. Cook

Master of Science candidate (JLR) and Professor of Forestry (JEC), College of Natural Resources, University of Wisconsin-Stevens Point, 800 Reserve St., Stevens Point, WI 54481.
JLR is the corresponding author: to contact, call 920-810-6103 or email at jralston@new.rr.com.

ABSTRACT

The Wisconsin Department of Natural Resources (WDNR) has designated barrens communities as high priority for restoration. These communities are globally imperiled and support many endangered and threatened species. Many barrens are in various stages of degradation or are unmanaged. Restoration of structure and fire may increase understory richness of true prairie species (Tester 1989). An intermediate term evaluation of management efforts at the Quincy Bluff and Wetlands Area (Quincy Bluff) demonstrated only small changes with harvesting and return of fire (Nielsen 2003). This discrepancy in results may reflect the length of fire suppression and the degree of degradation at Quincy Bluff. Prior to initiating management, fire had been absent from Quincy Bluff for approximately 70 years, while other restoration sites have reported fire suppression of approximately 30 years.

Quincy Bluff is located in Adams County, Wisconsin and consists of over 1500 ha of several community types, including considerable oak-pine barrens. The objectives of this study were to test the impact of prescribed fire, timber harvest, and the seed bank on understory plant diversity at Quincy Bluff. Thirty-nine permanent transects were established prior to initiation of restoration efforts in five of The Nature Conservancy (TNC) management units and in the unmanaged WDNR property. An additional 26 newly established, independent transects and the 39 previously established transects were monitored in 2010. The following were determined: cover and presence of all living vegetation <1 m tall; shrub and small tree cover and density; and tree cover, density, and basal area. Soil samples were obtained for greenhouse analysis of the seed bank. Species richness and ground layer cover in 2010 were lowest in management units treated with prescribed fire alone and were 50 percent to 150 percent greater in units with substantial reduction in canopy cover and basal area. Structural changes resulted from timber harvest and/or a tornado that struck one management unit in 2004. After a literature review, we generated a list of species likely to be indicators of barrens vegetation (Bray 1960, Curtis 1959, Will-Wolf and Stearns 1999). In 2010, these barrens indicator species had the largest percent cover in the control unit, however, cover of barrens indicator species declined in all units. In units with reduced canopy cover and basal area, barrens indicator species richness increased by two to three species and declined or remained the same in the other units. Frequent fires eliminated shrubs and small trees. Greenhouse analysis of the seed bank demonstrated low species richness and produced few germinates of any restoration value.

Barrens, if left unmanaged, degrade to closed canopy forests with low species richness and low ground layer cover. Frequent fires over 17 years were ineffective at reducing canopy cover or basal area, and understory species richness remained low. Manipulation of the tree structure through timber harvest and/or natural processes (i.e., tornado) increased understory species richness, however, species determined to indicate healthy barrens communities decreased in relative cover, and richness only increased marginally. These results confirm that the resiliency of this system may be exceeded by the long period without management and the seven decades of fire suppression.

LITERATURE CITED

Bray, J.R. 1960. **The composition of savanna vegetation in Wisconsin.** Ecology. 41: 721-732.

Curtis, J.T. 1959. **The vegetation of Wisconsin.** Madison, WI: The University of Wisconsin Press. 657 p.

Nielsen, S.; Kirschbaum, C.; Haney, A. 2003. **Restoration of midwest oak barrens: Structural manipulation or process-only?** Conservation Ecology. 7: 10-24.

Tester, J.R. 1989. **Effects of fire frequency on oak savanna in east-central Minnesota.** Bulletin of the Torrey Botanical Club. 116: 134-14.

Will-Wolf, S.; Stearns, F. 1999. **Dry soil oak savanna in the Great Lakes Region.** In: Anderson, R.C.; Fralish, J.S.; Baskin, J.M., eds. Savannas, barrens, and rock outcrop plant communities of North America. New York, NY: Cambridge University Press: 135-154.

PRESCRIBED FIRE AND THINNING IMPACTS ON FINE FUELS AT THE WILLIAM B. BANKHEAD NATIONAL FOREST, ALABAMA

Callie Jo Schweitzer and Stacy L. Clark

Research Forester (CJS), U.S. Forest Service, Southern Research Station,
P.O. Box 1568, Normal, AL 35762; Research Forester (SLC). CJS is corresponding author:
to contact call 256-372-4230 or email at cschweitzer@fs.fed.us.

INTRODUCTION

Prescribed burning and thinning are intermediate stand treatments whose consequences when applied in mixed pine-hardwood stands are unknown. The William B. Bankhead National Forest in north Alabama has undertaken these two options to restore unmanaged loblolly pine (*Pinus taeda* L.) plantations to hardwood-dominated stands. Our objective was to evaluate differences in fine fuel and duff dynamics in a combination of thinning and fire regimes.

METHODS

We installed a randomized complete block design with a 3 x 2 factorial treatment arrangement and four replications of each treatment. Treatments were three residual basal areas (50 ft^2 ac^{-1}, 75 ft^2 ac^{-1}, and an untreated control) with two burn frequencies (burns every 3 years and an unburned control). Two prescribed burns were implemented. We collected measurements before and after each treatment implementation. A 1 ft^2 wooden frame was used to collect samples, which were sorted into duff, leaves, 1-hour fuel (0.1-0.25 inches), 10-hour fuel (0.26-1 inch), fruit, and bark categories. Samples were oven dried until no change in weight was detected and then weighed to the nearest 0.01 g.

RESULTS

Main effect of thinning was significant for duff, 10-hour, 1-hour, and bark immediately after thinning. Compared to controls, thinning increased duff (+4.3 and +5.7 tons ac^{-1} for 50 ft^2 ac^{-1} and 75 ft^2 ac^{-1}, respectively), 1-hour (+0.2 tons ac^{-1} for both thinning treatments), 10-hour (+1.6 and +1.4 tons ac^{-1} for 50 ft^2 ac^{-1} and 75 ft^2 ac^{-1}, respectively), and bark loads (+0.4 and +0.3 tons ac^{-1} for 50 ft^2 ac^{-1} and 75 ft^2 ac^{-1}, respectively). Three years following thinning, only leaf litter had significant differences with a reduction of -1.1 and -0.8 tons ac^{-1} in 50 ft^2 ac^{-1} and 75 ft^2 ac^{-1} treatments, respectively. Burning and its interaction with thinning were not significant after the first burn. After the second burn, leaf litter decreased by 0.5 tons ac^{-1} compared to controls, and bark increased by 0.1 tons ac^{-1}. Leaf litter decreased by 0.5 tons ac^{-1} after the second burn, and bark increased by 0.1 tons ac^{-1}. Duff, 1-hour, 10-hour, and fruit were not affected by the first or second burn.

CONCLUSIONS

Burning alone and in conjunction with thinning as applied in this study had minimal effects on fine fuels and duff. Thinning produced more of a reduction in fine fuels than did burning, likely due to higher decomposition rates related to increased sunlight hitting the forest floor. Based on these results, we do not recommend using prescribed fire in these stands to reduce fine fuel and duff loads, but recognize fire has other benefits not measured in this study for vegetation diversity and wildlife habitat.

SIMULATING TREATMENT EFFECTS IN PINE-OAK FORESTS OF THE OUACHITA MOUNTAINS

Ayn Shlisky, Kori Blankenship, and Shawn M. Cochran

Regional Analyst (AS), U.S. Forest Service, Pacific Northwest Region, 333 SW. First Ave., Portland, OR 97204; Fire Ecologist (KB), The Nature Conservancy; Wildlife Biologist (SMC), U.S. Forest Service, Ouachita National Forest. AS is corresponding author: to contact, call 503-808-2284 or email at ajshlisky@fs.fed.us.

ABSTRACT

Effective land management decisionmaking depends on scientifically sound analyses of management alternatives relative to desired future conditions and environmental effects. We used a state-and-transition model to evaluate likely future landscape conditions in a pine-oak forest on the Ouachita National Forest, Arkansas based on current and potential future alternative management actions. Our objectives were to: (1) demonstrate the use of state-and-transition models in project planning, and (2) create simple "what if" scenarios to supplement the project environmental impact assessment and facilitate more informed decisionmaking through relative comparisons. We used the model to simulate and compare the effects of several management alternatives:

A. Current Management
B. Regeneration Harvest/Thinning
C. Woodland Management
D. Regeneration Harvest/Thinning + Climate Change
E. Woodland Management + Climate Change

The effects of these alternatives were also compared against modeled ecological reference conditions.

At the time of the study, a national forest interdisciplinary team was completing a project-level environmental assessment of alternative management scenarios across the 16,700 acre Lower Irons Fork/Johnson Creek watershed. The watershed is located within the Ouachita National Forest near the town of Mena in western Arkansas. It is comprised primarily of pine-oak forest and woodland and has a history of active fire management. The watershed is a drinking water source for the town of Mena, Arkansas, offers recreational opportunities including hunting and fishing, and is home to two federally-endangered species: the red-cockaded woodpecker (*Picoides borealis)* and the harperella plant (*Ptilimnium nodosumis*).

We modified the LANDFIRE Ozark-Ouachita Shortleaf Pine-Oak Forest and Woodland model (USDA FS and USDI 2009) using stand exam and other forest data to represent current landscape structure and disturbance probabilities. Timber harvest volume per acre coefficients were estimated from a similar nearby project, and smoke production values for particulate matter (PM 10 and 2.5) were estimated from the First Order Fire Effects Model. Forest colleagues and U.S. Forest Service Southern Research Station scientists provided peer review. We used the Vegetation Dynamics Development Tool and the Path Landscape Model, a state-and-transition modeling framework and platform, to simulate the effects of the various alternatives after 10 years.

Our results indicate that a woodland management emphasis generally yielded landscape structure and fire frequencies closer to the desired future condition specified in the 2005 Ouachita National Forest Revised Forest Plan (USDA FS 2005) compared to a regeneration harvest/thinning emphasis (Figs. 1 and 2). When potential climate effects were considered, the woodland management emphasis also yielded greater smoke (Fig. 3) and woody biomass harvest outputs (Fig. 4) than the regeneration harvest/thinning emphasis.

These findings suggest that Forest Plan revisions should reevaluate the desired future conditions for pine-oak forest in light of the fact that it does not currently include a standard for mid-seral forest structure, and that the existing desired future condition standard for late seral open woodland is lower than LANDFIRE reference conditions. While the model outputs have proven to be useful, the process forced the team to test assumptions and document knowledge, two intangible but valuable outcomes.

LITERATURE CITED

U.S. Department of Agriculture, Forest Service; U.S. Department of Interior. 2009. **LANDFIRE 1.1.0 Vegetation Dynamics Models [Database].** Available at http://www.landfire.gov. (Accessed 11 August 2009).

U.S. Department of Agriculture, Forest Service. 2005. **Revised land and resource management plan: Ouachita National Forest-Arkansas and Oklahoma.** Management Bulletin R8-MB 124A. Washington, DC.

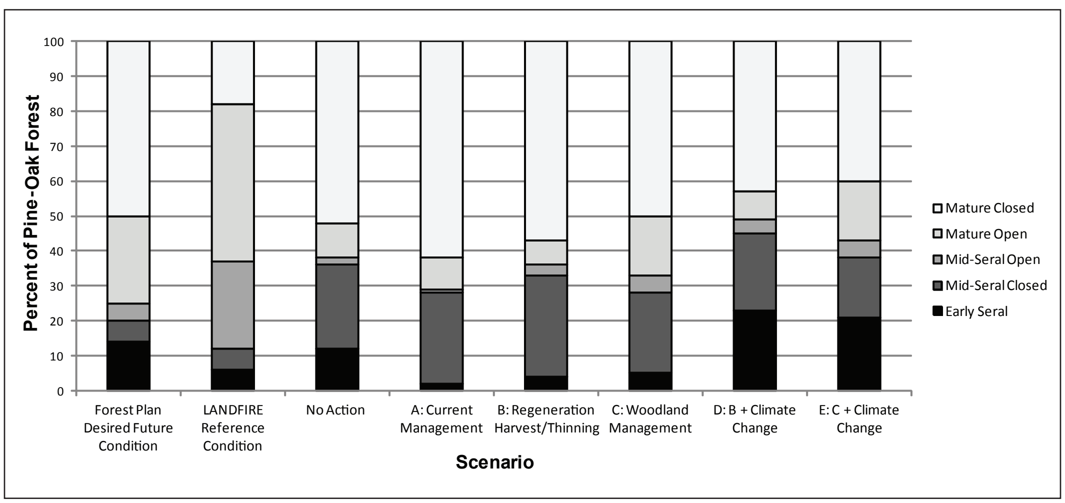

Figure 1.—Proportions of pine-oak forest and woodland structural stages resulting from modeled management alternatives, ecological reference conditions, and revised forest plan desired conditions after a 10-year simulation.

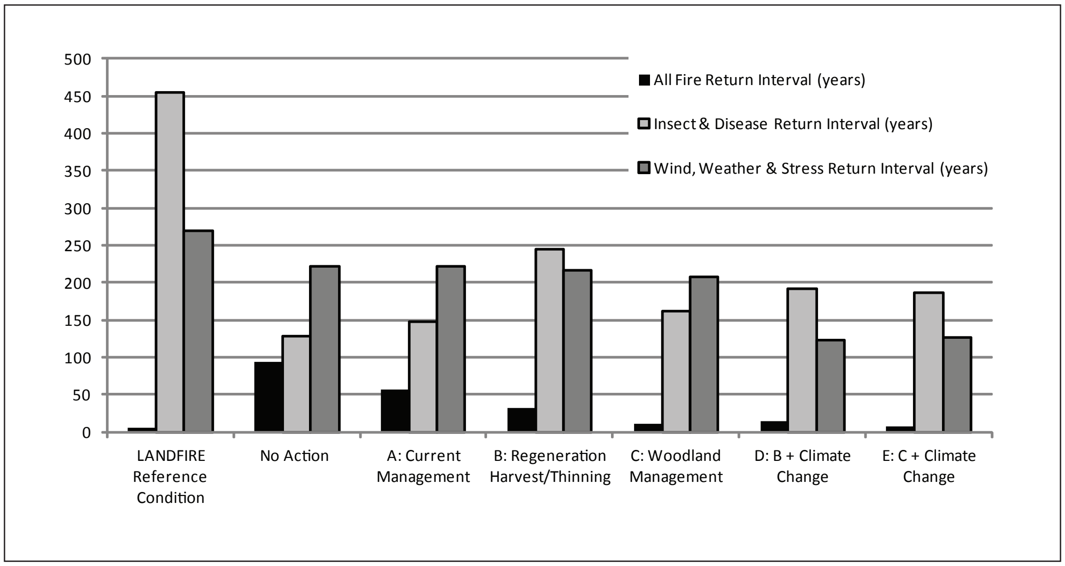

Figure 2.—Comparative return intervals for fire, insects and disease, and wind/weather/stress disturbances resulting from modeled management alternatives and ecological reference conditions.

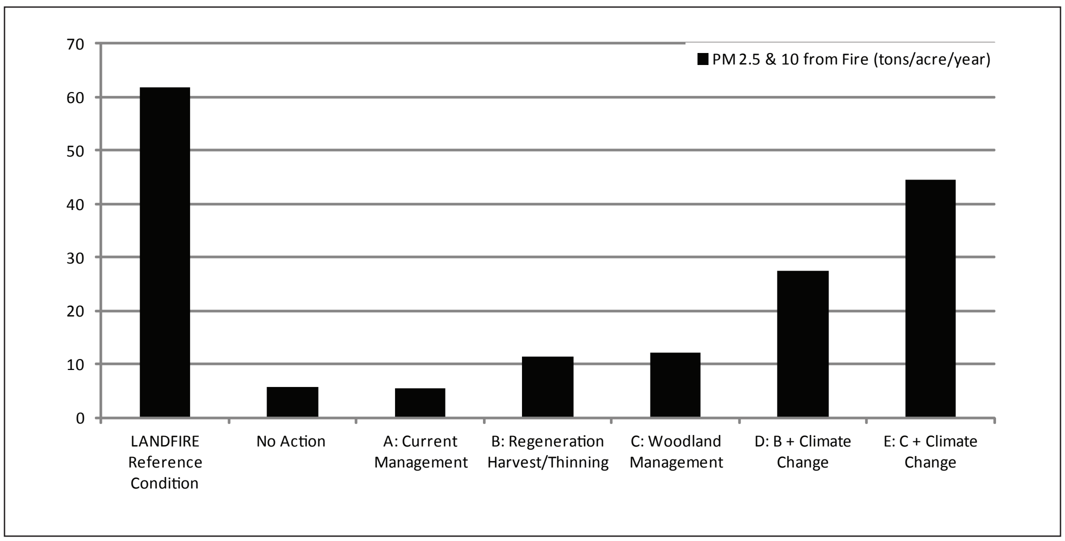

Figure 3.—Comparative smoke emissions resulting from modeled management alternatives and ecological reference conditions.

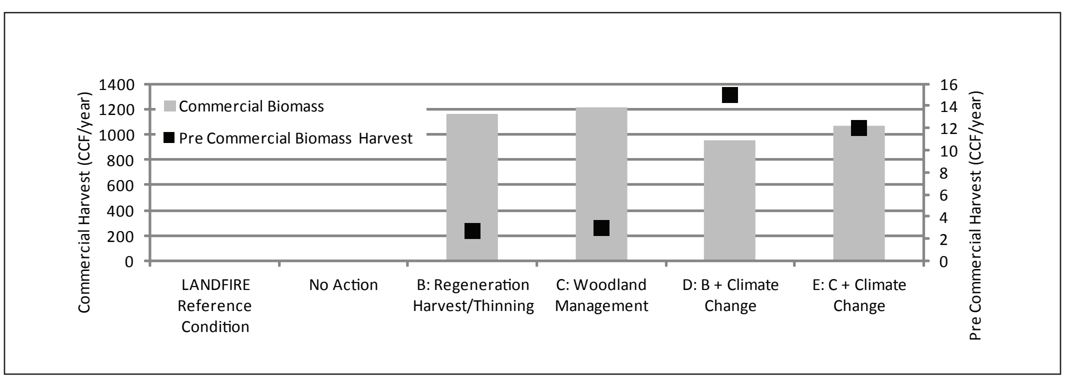

Figure 4.—Comparative biomass harvest resulting from modeled management alternatives.

VEGETATION STRUCTURE AND SOIL WATER CONTENT AFTER 60 YEARS OF REPEATED PRESCRIBED BURNING IN THE MISSOURI OZARKS

Kirsten Stephan and Jason A. Hubbart

Assistant Professor of Biology (KS), Department of Life and Physical Sciences, Lincoln University of Missouri, 816 Chestnut St., Jefferson City, MO 65101; Assistant Professor of Forest Hydrology (JAH). KS is corresponding author: to contact, call 573-681-5116 or email at stephank@lincolnu.edu.

INTRODUCTION

Two important interrelated factors determining ecosystem structure and function are vegetation composition and soil moisture. Both are affected directly and indirectly by fire (Hart and others 2005). Few studies have characterized the long-term effects of repeated spring prescribed burning on these ecosystem characteristics. As an exception, Lewis and Harshbarger (1976) found that 20 years of repeated prescribed burning had significant and consistent effects on shrubs and herbs. However, post-fire soil moisture has been shown to increase (Klock and Helvey 1976), decrease (Campbell and others 1977), or remain the same (Wells and others 1979) after single fire events. This could be attributed to the timing of measurements, sampling design, and/or differential fire effects on the many factors that influence soil moisture, such as interception by plants or forest floor, evaporation from the canopy or the soil, transpiration, and infiltration rates.

The objectives of this study were to quantify the effects of 60 years of repeated spring prescribed burning at annual and 4-year intervals on a) vegetation composition and structure, b) soil physical properties, and c) their effects on soil moisture in a Missouri oak-hickory forest.

STUDY AREA

The research was conducted at University Forest Conservation Area located in the Ozark Mountains near Wappapello in southeast Missouri. Data were collected from a total of six 40 m by 40 m plots in two sites. Site 1 and site 2 are approximately 3.2 km apart and were established in 1949 and 1951, respectively. At each site, plots are separated by a 10 m buffer. At each site, one of the studied plots is burned every spring (annual burn treatment, A), every 4 years (periodic burn treatment, P), or never (control, C). The last periodic burns prior to data collection took place in 2007 (site 2) and 2009 (site 1).

METHODS

Vegetation cover and forest floor mass data were collected in 2010. Cover of herbaceous and woody understory vegetation (<0.5 m height) and woody midstory vegetation (0.5-2 m height) was assessed using 32 sampling quadrats (1 m by 1 m) per plot. Quadrats were placed every 5 m along transects spaced 10 m apart. Cover was estimated to the nearest percent. Cover of shrubs and trees in the midstory and overstory >1.4 m was assessed via the Leaf Area Index (LAI) using a ceptometer in nine locations per plot.

Forest floor mass was quantified by collecting material from eight circular areas (30 cm diameter) per plot. Forest floor material was air-dried; woody and nonwoody components were separated and weighed.

Soil physical properties were assessed in eight locations per plot in 2009. Infiltration capacity was measured via double-ring infiltrometers. Bulk density was determined from soil cores (5 cm diameter, 5 cm long). Soil volumetric water content (VWC) was continuously measured at 15 cm depth in one location per plot at 30 minute intervals from 2008 to 2010.

RESULTS & DISCUSSION

Tree and shrub cover (>1.4 m) was highest in the controls (LAI: A 2.3, P 2.1, C 3.4). This was attributed to fire-related tree mortality at the beginning of the study in annual and periodic burns. We predicted that interception, canopy evaporation, and transpiration losses by midstory and overstory trees lowered soil moisture in the controls more strongly relative to other treatments.

Cover of understory plants was highest in annual burns (cover in percent: A 40.0, P 37.2, C 14.0); cover of understory and midstory (<2 m) plants together was highest in the periodic burns (cover in percent: A 40.0, P 64.5, C 31.0). This was due to lack of woody seedling survival in annual burns but abundant resprouting in periodic burns. We predicted that interception, canopy evaporation, and transpiration losses by these plants reduced soil moisture in the periodic burn more strongly than in other treatments.

Forest floor (mainly leaf litter) was least abundant in annual burns because of frequent combustion (forest floor content in t ha^{-1}: A 4.7, P 8.5, C 15.4). The predicted effect on soil moisture was ambiguous. On one hand, low interception by and evaporation from a thin litter layer would facilitate precipitation to enter into the mineral soil. On the other hand, once in mineral soil, water is more likely to evaporate from a soil lacking litter insulation.

Infiltration capacity was lowest in annual burns (A 11, P 41, C 107 cm h^{-1}). This was likely due to a compacted top layer of soils from raindrop impact on bare ground in annual burns (bulk density in g cm^{-3}: A 0.99, P 0.96, C 0.94). We predicted that lower infiltration capacity decreased soil moisture in annual burns relative to other treatments.

In all treatments, direct measurements of soil moisture showed the expected temporal pattern of decreased soil moisture during the growing seasons due to increased evapotranspiration. However, there was no consistent fire effect on soil moisture. While VWC in annual burns was intermediate to the other treatments at both sites, VWC of the control was highest in site 1 but lowest in site 2 for the majority of the measurement period. This indicates the existence of significant microsite differences between sensor locations, masking any potential fire effect.

CONCLUSIONS

While burn treatments affected vegetation and soil physical characteristics in a similar way in both sites, there was no consistent treatment effect on soil volumetric water content between the two sites. We hypothesize that this is likely due to large microsite variability commonly observed in soils. Several VWC probes per plot would be required to capture plot-scale soil moisture levels. Despite the low intensity of spring prescribed burning, repeated burning over decades has dramatic effects on vegetation characteristics and soil physical properties. A mechanistic understanding of the effects of long-term prescribed burning will result in better informed forest management decisions in Ozark oak-hickory forest.

ACKNOWLEDGMENTS

We thank Staria Vanderpool (plant data collection and identification), Edward Bulliner, Graham Freeman and Thomas Stokely (LAI, soil sampling), Brittany Winston (forest floor processing), and Marie Obourn, UM Forest Manager. Financial support was provided by DOD (W911NF-05-2-0003), John Yang at LU Cooperative Research, USDA CSREES/NIFA McIntire-Stennis (Accession #:218480), and the University of Missouri.

LITERATURE CITED

Campbell, R.; Baker, M., Jr.; Folliott, P.; Larson, F.; Avery, C. 1977. **Wildfire effects on a ponderosa pine ecosystem: an Arizona case study.** Res. Pap. RM-191. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 12 p.

Hart, S.C.; DeLuca, T.H.; Newman, G.S.; MacKenzie, M.D.; Boyle, S.I. 2005. **Post-fire vegetative dynamics as drivers of microbial community structure and function in forest soils.** Forest Ecology and Management. 220: 166-184.

Klock, G.; Helvey, J. 1976. **Soil water trends following wildfire in the Entiat Experimental Forest.** In: Komarek, E.V., ed. Proceedings, 15th Tall Timbers Fire Ecology Conference, Pacific Northwest; 1974 October 16-17; Portland, OR. Tallahassee, FL: Tall Timbers Research Station: 193-200.

Lewis, C.; Harshbarger, T. 1976. **Shrub and herbaceous vegetation after 20 years of prescribed burning in the South Carolina coastal plain.** Journal of Range Management. 29: 13-18.

Wells, C.; Campbell, R.; DeBano, L.F.; Lewis, C.; Fredericksen, R.; Froelich, R.; Dunn, P. 1979. **Effects of fire on soil: a state of the art knowledge review.** Gen. Tech. Rep. WO-7. Washington, DC: U.S. Department of Agriculture, Forest Service. 34 p.