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Woodland salamander response to two prescribed fires in the central Appalachians

W. Mark Ford[a,*], Jane L. Rodrigue[b], Ella L. Rowan[c], Steven B. Castleberry[d], Thomas M. Schuler[b]

[a] U.S. Army Engineer Research and Development Center, Environmental Lab, Vicksburg, MS 39180-6199, United States
[b] U.S. Forest Service, Northern Research Station, Parsons, WV 26287, United States
[c] Washington Department of Fish and Wildlife, Spokane Valley, WA 99126, United States
[d] Warnell School of Forestry and Natural Resources, Athens, GA 30602, United States

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Using coverboard arrays, we monitored woodland salamanders on the Fernow Experimental Forest in the central Appalachian Mountains, West Virginia, USA prior to and following two prescribed fires in mixed oak (*Quercus* spp.) forest stands. Treatments were burn plots on upper slopes or lower slopes fenced to prevent white-tailed deer (*Odocoileus virginianus*) herbivory or control plots that were unfenced and unburned. Most of the 7 species we observed were the mountain dusky salamander (*Desmognathus ocropheaus*), red-backed salamander (*Plethodon cinereus*) and slimy salamander (*Plethodon glutinosis*). Significant population responses were difficult to interpret with numerous treatment and year interactions. Results largely were equivocal. We found no change in woodland salamander assemblage prior to burning or afterwards. There were few differences in adult to juvenile ratios of salamanders among treatments. Still, *a priori* contrasts of mountain dusky salamanders and red-backed salamander counts corrected for detection probability were greater under coverboards in the 2 years monitored after both prescribed fires had occurred than before burning or in unburned controls. This suggests that these species responded to the reduced leaf litter on the forest floor by utilizing coverboards more. Similarly, the three predominate species of salamanders also were more numerous under coverboards in plots subjected to deer herbivory with less subsequent forest floor vegetation as compared to those burned plots that were fenced. Our observations would suggest that woodland salamanders somewhat are tolerant of two prescribed fires within close temporal proximity. However, because woodland salamanders can be significantly reduced following timber harvest, continued research is needed to fully understand impacts of fire as a pre-harvest management tool in central Appalachian forests.

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1. Introduction

Coincident with post-Pleistocene climate change and use of fire by native Americans, oak (*Quercus*)-dominated forests have been present throughout eastern North America for the past 10,000+ years (Delcourt et al., 1998). In the central Appalachians, decades of fire suppression and other ecological factors have favored the establishment of shade-tolerant species with a concomitant paucity of oak regeneration (Brose et al., 2006; Nowacki and Abrams, 2008). This shift in woody species assemblage along with white-tailed deer (*Odocoileus virginianus*) herbivory prevents oak reestablishment following timber harvest, or after natural mortality of canopy dominant trees (Schuler, 2004; Brose et al., 2006; Miller et al., 2009). Attempts at reintroducing fire pre-harvest to kill competing vegetation have met with varied success (Brose et al., 2006). A single low-intensity prescribed fire generally will fail to alter under- and mid-story conditions enough to increase oak establishment (Barnes and Van Lear, 1998; Apsley and McCarthy, 2004). Conversely, multiple burns can reduce shade-tolerant competition and open the mid-story that coincident with good acorn crops can increase oak seedlings in the regeneration pool (Brose et al., 2006).

Fire reintroduction, particularly repeated burning in forests that have developed over decades with fire suppression raises questions about impacts to forest floor dwelling vertebrates (Ford et al., 1999; Rowan et al., 2005). Central Appalachian forests are noted for their high biodiversity, and managers often are required to consider management impacts across a broad array of taxa for which biological information often is lacking (Keyser and Ford, 2006). Woodland salamanders (Family Plethodontidae) represent one such group of high conservation concern, comprising a large portion of the regional vertebrate biomass of forests (Pauley et al., 2006). Woodland salamanders are lungless and require moist

* Corresponding author. Tel.: +1 601 634 4602; fax: +1 601 634 3726.
E-mail address: mfordfs@yahoo.com (W.M. Ford).

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conditions to allow for dermal respiration, therefore disturbances that reduce extant leaf litter, leaf litter inputs, cover objects, or otherwise open the forest floor to more light and higher forest floor temperatures are believed detrimental (Maerz et al., 2009). In both the central and southern Appalachians, impacts of timber harvest on woodland salamanders are highest following canopy removal after clearcutting—due to drastic reductions in litter input (Petranka et al., 1993; Homyack and Haas, 2009). Recovery periods require several years post-harvest (Ash, 1997; Ford et al., 2002a,b). Systems retaining some overstory experience lesser or no measurable decline (Ford et al., 2000; Homyack and Haas, 2009).

Although a wealth of research on woodland salamander response to timber harvesting exists, data on woodland salamander response to prescribed burning in deciduous forests in general and Appalachian forests in particular is limited (Pilliod et al., 2003; Renken, 2006). There are data on herpetofaunal response to fire from Southeastern pine-dominated systems where natural and anthropogenic fire is a frequent and necessary sustaining disturbance agent (Russell et al., 1999). Still, results from single-application prescribed burns in deciduous forest communities in the eastern United State suggest that impacts to woodland salamanders were negligible (Dechant, 2007; Renken, 2006). In part, these equivocal effects may be due to low sample sizes or non-robust observational designs for some research (Kirkland et al., 1996; Keyser et al., 2004). Short-term impacts to woodland salamanders in the southern Appalachians of western North Carolina following single applications of fire appeared to have been negligible (Ford et al., 1999; Greenberg and Waldrop, 2008). The distribution and abundance of woodland salamanders was related mostly to slope position and overall site quality rather than burn impact (Ford et al., 1999).

Similar to the concept that multiple stand entries for partial timber harvest regimes may negatively impact salamander communities cumulatively over time (Semlitch et al., 2007; Moseley et al., 2008), fire-impacts might take longer periods to manifest themselves or that multiple fires might cross a disturbance threshold that could impact woodland salamanders (Greenberg and Waldrop, 2008). If multiple fires over a relatively short time prior to timber harvest are required to facilitate oak regeneration in future stands in the central Appalachians, managers need more information to weigh the relative risks to taxa of high conservation concern such as woodland salamanders and may need to devise mitigative strategies to lessen impacts. Thus, our objective was to track woodland salamander abundance by species prior to and after the application of two consecutive prescribed fires in a central Appalachian mixed oak/Allegheny hardwood forest association.

2. Methods

We conducted our research in the Canoe Run Watershed on the Fernow Experimental Forest (39.03°N, 79.67°W), Tucker County in east-central West Virginia, USA. Located within the Allegheny Mountain portion of the Appalachian Plateau Physiographic Province, the 1900-ha area is managed by the USDA Forest Service Northern Research Station primarily to conduct long-term silvicultural and hydrologic research. The growing season is approximately 145 days (May–October) with 143 cm of annual precipitation. Because precipitation is evenly distributed throughout the year and summer temperatures are moderate (Pan et al., 1997), moisture deficits in the growing season are uncommon (Leathers et al., 2000). Elevations ranged from 615 to 800 m. Aspects were generally west-facing and the topography was varied from steep side slopes to gentle broad ridge-tops. Soils predominately were in the Calvin and Dekalb series derived from underlying sandstone and shale geology (Adams et al., 2008).

Though most of the Fernow Experimental Forest's overstory composition is best described as a mixed-mesophytic type of the Central Appalachian Broadleaf Forest (Braun, 1950; McNab and Avers, 1994), the overstory dominants at our study site were northern red oak (*Quercus rubra*), chestnut oak (*Q. prinus*), and white oak (*Q. alba*) in descending order of importance as measured by basal area (see Rowan et al., 2005 for a more detailed description). Forest stands probably regenerated in the years following the initial logging of the virgin forest in the area that is now the Fernow Experimental Forest in the early 20th Century (Schuler, 2004). The upper slope and ridgetop portion of the study area had been variably thinned up to 60 percent overstory stocking in 1983 whereas lower slope forests were fully stocked (Schuler and Miller, 1995.

We established 24 0.20 ha plots (10 lower slopes and 14 upper slopes) where we burned 20 of the 24 plots with two prescribed fires just prior to the growing seasons in 2002 (lower slope) and 2003 (upper slope) and in 2005 (both slopes) (Schuler et al., 2010). Two plots on the upper and 2 on lower slopes remained as unburned controls throughout the duration of our study. The first prescribed fire in April 2002 was interrupted because of poor weather conditions allowing us to only burn the lower slope. We burned the upper slope the following spring. We then treated the entire area with a second prescribed fire in April 2005, whereby all parts of the study site had then been burned for a second time. We burned sites using the strip head fire technique ignited with hand-held drip torches. In general, fire behavior was mostly moderate or low intensity during all of our prescribed burns (i.e., flame lengths less than 1 m from the combustion of leaf litter and 1-h surface fuels). However, actual fire spread rates were greater than those predicted by modeling but were similar to those reported in southeastern Ohio (Iverson et al., 2004). Further, as part of ongoing vegetation research (Schuler et al., 2010), 10 of the 20 burn plots (4 lower slope and 6 upper slope) also were fenced to prevent white-tailed deer herbivory.

Using coverboard arrays (DeGraaf and Yamasaki, 2002), we surveyed woodland salamanders monthly from April to October from 2001 to 2005 and in the months of May, July and September in 2006 and 2007. We conducted surveys after precipitation events whenever possible with monthly surveys separated by 3–5 weeks. We placed coverboards in the study site in 2000 to allow them to “weather” for 6 months prior to our first salamander surveys. Coverboard arrays consisted of 1 m^2 plywood cut into 9 identical square pieces (1.27 cm in thickness) and placed flush on the forest floor adjacent to each other (following Pauley, 1995). Within each 0.20 ha plot, we placed three coverboard arrays located along the central axis of the plot, separated by approximately 10–15 m. Upon capture, we identified each salamander to species, recorded mass and snout-vent length (SVL). To avoid double-counting individuals when multiple salamanders were present within an array, we did not return salamanders until all individuals had been weighed and measured. We classified salamanders as adults if maximum SVL's exceeded values reported for juveniles elsewhere (Petranka, 1998; Homyack and Haas, 2009).

To test the effect of prescribed burning on the relative abundance of woodland salamanders within our repeated measures two-factor crossed before-after/control-treatment design, we fit generalized linear mixed models (PROC GLIMMIX; SAS 9.3, SAS Inc., Cary, NC) in a two-factor design with interactions and repeated measures to count data from coverboards by individual species as by coverboard array. We considered sample year and 6 plot type/slope treatment combinations as main effects: upper slope deer fenced, upper slope unfenced, lower slope deer fenced, lower slope unfenced, and unburned and unfenced controls on upper slopes and lower slopes. Also, we considered months within years as our repeated measure to account for the intra-seasonal variation in above ground activity among woodland salamander species in the Appalachians and the possible sampling bias thereof (Moore et