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Woodland salamander response to two prescribed fires in the central Appalachians

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ABSTRACT

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.

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

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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

Table 1

Detection probabilities for woodland salamanders using coverboard sampling (burn plots at 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), Fernow Experimental Forest, West Virginia, 2001–2007. Overall detection probabilities were calculated from the seasonal occupancy and detection model (see text) and used as a correction factor for woodland salamander coverboard counts.

	Mtn. dusky salamander		Red-backed salamander		Slimy salamander	
Treatment	ρ	SE	ρ	SE	ρ	SE
Upper slope fenced	0.19	0.07	0.55	3.01	0.20	0.04
Upper slope unfenced	0.38	0.03	0.53	0.02	0.15	0.04
Lower slope fenced	0.42	0.04	0.33	0.03	0.17	0.03
Lower slope unfenced	0.15	0.04	0.27	0.03	0.29	1.08
Upper slope control	0.39	0.05	0.45	1.79	0.25	0.11
Lower slope control	0.25	0.05	0.40	1.64	0.11	0.03

Table 2

Mean captures of woodland salamanders (burn plots at 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) by coverboard array, Fernow Experimental Forest, West Virginia, 2001–2007 (see text for details).

	Mtn. dusky salamander		Red-backed salamander		Slimy salamander	
Treatment	Mean	SE	Mean	SE	Mean	SE
2001						
Upper slope fenced	0.02	0.02	1.75	0.19	0.28	0.12
Upper slope unfenced	1.24	0.32	1.58	0.18	0.03	0.03
Lower slope fenced	1.27	0.34	1.18	0.31	0.84	0.22
Lower slope unfenced	0.33	0.12	0.89	0.23	1.53	0.40
Upper slope control	0.56	0.37	2.07	0.38	0.26	0.26
Lower slope control	0.28	0.14	1.88	0.43	0.43	0.21
2002						
Upper slope fenced	0.07	0.04	1.17	0.15	0.40	0.14
Upper slope unfenced	1.30	0.31	1.21	0.18	0.06	0.05
Lower slope fenced	2.05	0.48	0.81	0.19	0.28	0.14
Lower slope unfenced	0.47	0.14	0.78	0.18	1.08	1.36
Upper slope control	0.79	0.35	1.40	0.31	0.77	0.43
Lower slope control	0.43	0.16	1.46	0.46	0.44	0.21
2003						
Upper slope fenced	0.24	0.12	2.09	0.43	0.09	0.09
Upper slope unfenced	2.23	0.37	1.99	0.19	0.39	0.11
Lower slope fenced	1.15	0.30	0.73	0.17	0.72	0.22
Lower slope unfenced	0.12	0.06	0.67	0.16	2.24	0.42
Upper slope control	1.44	0.48	1.26	0.27	0.00	0.00
Lower slope control	0.24	0.12	2.09	0.43	0.09	0.09
2004						
Upper slope fenced	0.09	0.04	1.21	0.15	0.43	0.14
Upper slope unfenced	1.54	0.30	1.31	0.14	0.36	0.09
Lower slope fenced	0.97	0.22	0.73	0.20	0.78	0.22
Lower slope unfenced	0.22	0.08	0.63	0.16	2.32	0.41
Upper slope control	1.84	0.44	1.47	0.29	0.44	0.30
Lower slope control	0.37	0.14	1.02	0.30	0.75	0.33
2005						
Upper slope fenced	0.06	0.04	3.17	0.34	0.84	0.26
Upper slope unfenced	2.20	0.44	2.91	0.32	0.21	0.09
Lower slope fenced	2.93	0.77	1.24	0.31	0.94	0.28
Lower slope unfenced	0.39	0.15	2.38	0.47	2.60	0.52
Upper slope control	2.02	0.44	1.57	0.41	0.33	0.33
Lower slope control	0.77	0.33	1.00	0.25	0.78	0.34
2006						
Upper slope fenced	0.05	0.05	1.31	0.22	1.73	0.47
Upper slope unfenced	0.93	0.36	0.97	0.27	0.33	0.16
Lower slope fenced	1.70	0.61	1.22	0.34	0.67	0.32
Lower slope unfenced	0.47	0.19	0.13	0.13	2.78	0.85
Upper slope control	1.74	0.69	0.94	0.38	0.00	0.00
Lower slope control	0.64	0.37	2.51	0.79	0.49	0.33
2007						
Upper slope fenced	0.00	0.00	1.15	0.23	1.01	0.40
Upper slope unfenced	0.69	0.41	0.49	0.15	0.63	0.22
Lower slope fenced	0.57	0.27	1.02	0.36	0.98	0.39
Lower slope unfenced	0.22	0.12	0.64	0.24	1.48	0.60
Upper slope control	0.94	0.49	1.01	0.47	1.41	0.98
Lower slope control	0.76	0.37	1.62	0.52	0.23	0.23

al., 2001; Bailey et al., 2004). For these models, we used logarithmic link functions with compound-symmetry structure because the count data displayed a negative binomial distribution and the main effects were random rather than fixed.

Prior to analysis, we corrected woodland salamander count data for each species examined by dividing count data with overall species-specific detection probabilities (Anderson, 2003; Cruickshank, 2004) calculated for each type/slope treatment combination to reduce biases associated with higher and lower woodland salamander detections in disturbed and undisturbed habitats, respectively (Cummer and Painer, 2007). We used program PRESENCE (Hines and McKenzie, 2008) to calculate detection probabilities using the seasonal occupancy and detection model $\psi(.)$, $\gamma(.)$, $\varepsilon = 1- \gamma,\rho(.)$. We used pre-planned orthogonal contrasts (SAS 9.3, SAS Inc., Cary, NC) to compare salamander counts between: control plots vs. burn plots in 2001 prior to any fire, fenced vs. unfenced in burn plots after 2001, upper slopes vs. lower slopes in burn plots after 2001, upper slopes vs. lower slopes in control plots after 2001, burn plots vs. control plots in years with burning, and burn plots vs. control plots in 1- and 2-years following the second burning. We used Fisher's Exact tests (SAS 9.3, SAS Inc., Cary, NC) to examine if the distribution of juvenile and adult woodland salamanders by species were equitably distributed across type/slope combinations by years.

3. Results

From 2001 through 2007, we found 1665 red-backed salamanders (*Plethodon cinereus*), 455 mountain dusky salamanders (*Desmognathus ocropheaus*), 331 slimy salamanders (*P. glutinosis*), 13 red-spotted newts (*Notophthalmus viridescens*), 12 northern two-lined salamanders (*Eurycea bislineata*), 2 dusky salamanders (*D. fuscus*), and one each of Wehrle's salamander (*P. wehrlei*) and northern red salamander (*Pseudotrition ruber*). For the three species present in large enough numbers to permit statistical analysis, overall monthly detection probabilities were relatively low and highly variable across treatment categories (Table 1). Mean numbers of mountain dusky salamanders, red-backed salamanders and slimy salamanders at coverboard arrays also were highly variable across treatments over survey years (Table 2).

Year and treatment main effects were significantly different among numbers of red-backed salamanders, and slimy salamanders (Table 3). Year × treatment interactions also were significant for those species values (Table 3). For mountain dusky salamanders, only treatment main effects were significantly different (Table 3). Our *a priori* contrasts showed that mountain dusky and red-backed salamanders occurred in greater number in plots that had been burned twice as opposed to unburned plots (Table 4). Results largely were equivocal for slimy salamanders (Table 4). All three species were observed less in fenced plots than unfenced plots after the fires (Table 4). Slope effects also were variable, with more mountain dusky salamanders occurring in burned lower slopes and more red-backed salamander occurring in burned upper slopes (Table 4). Across all years and treatments, distribution of juvenile mountain dusky salamanders (SVL < 30 mm), juvenile red-backed salamanders (SVL < 34 mm) and juvenile slimy salamanders (SVL < 58 mm) versus adults for those species were largely equitable in distribution (Fisher's Exact tests $P > 0.1$). As exceptions, observed numbers were dissimilar from expected numbers for four salamander/year combinations: mountain dusky salamanders in 2005 (Fisher's Exact test $P = 0.027$), red-backed salamanders in 2001 (Fisher's Exact Test $P = 0.043$), 2005 (Fisher's Exact test $P = 0.0002$) and 2007 (Fisher's Exact test $P = 0.002$).

Table 3
Type III tests of year and treatment (burn plots at 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) effects for woodland salamander counts, Fernow Experimental Forest, West Virginia, 2001–2007.

Species	Effect	*d.f.*	*F*-value	*P*
Mtn. dusky salamander	Year	6, 238	1.63	0.1402
	Treatment	14.73	5, 2490	<0.0001
	Year × Treat.	28, 2490	1.21	0.2044
Red-backed salamander	Year	6, 117	2.42	0.0308
	Treatment	5, 2508	13.60	<0.0001
	Year × Treat.	30, 2205	2.76	<0.0001
Slimy salamander	Year	6, 353	3.19	0.0047
	Treatment	5, 2477	18.20	<0.0001
	Year × Treat.	28, 2477	1.50	0.0452

4. Discussion

Seven years of monitoring coverboards on the Fernow Experimental Forest demonstrated high variability in counts yet negligible demonstrable implications, if any, to two early spring prescribed fires in close temporal proximity on woodland salamander abundance. Our results largely were equivocal and sometimes

Table 4
A priori contrasts of treatment main effects combinations for woodland salamander counts, Fernow Experimental Forest, West Virginia, 2001-2007. Prescribed burns occurred in 2002 or 2003 and 2005 (see text for details).

	Contrast	*d.f.*	*F*-value	*P*
Mtn. dusky salamander	Burn = control (in 2001)	1, 2490	0.67	0.4134
	Fenced < unfenced (after 2001 in burn)	1, 2490	3.88	0.0490
	Burn upper slope < lower (after 2001)	1, 2490	11.33	0.0008
	Control upper slope > lower (after 2001)	1, 2490	4.65	0.0312
	Burn > control (in fire years)	1, 2490	7.54	0.0061
	Burn > control (after 2005)	1, 2490	8.00	0.0047
Red-backed salamander	Burn < control (in 2001)	1, 1681	5.52	0.0189
	Fenced < unfenced (after 2001 in burn)	1, 2531	12.18	0.0005
	Burn upper slope > lower (after 2001)	1, 2531	44.26	<0.0001
	Control upper slope = lower (after 2001)	1, 1945	0.84	0.3587
	Burn = control (in fire years)	1, 2017	1.94	0.1641
	Burn > control (after 2005)	1, 2440	10.27	0.0014
Slimy salamander	Burn = control (in 2001)	1, 2477	0.05	0.8180
	Fenced < unfenced (after 2001 in burn)	1, 2477	4.54	0.0332
	Burn upper slope < lower (after 2001)	1, 2477	31.81	<0.0001
	Control upper slope = lower (after 2001)	1, 2477	2.77	0.0959
	Burn = control (in fire years)	1, 2477	0.70	0.4019
	Burn = control (after 2005)	1, 2477	1.48	0.2236

counter-intuitive with year and treatment interactions along with low overall detection probabilities limiting the strength of our inferences, Regardless of slope position or protection from white-tailed deer herbivory, the extant woodland salamander assemblage consisting largely of mountain dusky, red-backed and slimy salamanders remained intact after the application of the two fires. We did not witness the cumulative impacts from repeated fire reducing woodland salamander abundance as Greenberg and Waldrop (2008) speculated might occur or the deleterious impact seen with repeated stand entry for timber harvest over successive decades (Homyack and Haas 2008; Moseley et al., 2008). Overall relative abundance of the three woodland salamanders species most observed were similar within treatment designations in 2007 at the study conclusion to those values in 2001 prior to any burning. This, combined with most prescribed fire applications in the Appalachians typically conducted at the relatively small stand or multiple-stand levels (Keyser and Ford, 2006; Lanham et al., 2006), means that managers could utilize burning without impacting woodland salamander viability across the wider landscape similar to the southern Appalachians (Ford et al., 1999; Greenberg, 2001) and adjacent upper Piedmont (Keyser et al., 2004). Response of woodland salamanders to one-time prescribed fires in the southern Appalachians (Ford et al., 1999; Greenberg and Waldrop, 2008) suggests no evidence for micro-habitat degradation. Still, we did observe higher mountain dusky salamander and red-backed salamander abundances under coverboards in burned rather than unburned plots after both fires and for red-backed salamanders following the second burn. In the growing seasons immediately following burning, there was noticeably reduced leaf litter (Schuler et al., 2010), possibly causing our coverboards to be utilized more by salamanders. Clumping behavior under residual cover objects following disturbance to forest floor habitats has been reported for woodland salamanders in other systems (Cummer and Painer, 2007; McKenny et al., 2006). We observed that mid-story tree and shrub mortality of species such as red maple and American beech continued in a delayed fashion following both fires. This probably continued to further reduce leaf litter inputs in subsequent survey seasons that would have been available for cover normally. Reduced mid-story and shrub layers may increase the amount of light reaching the forest floor enough to degrade (dry) the quality or extent moist micro-habitats utilized by woodland salamanders (Ash, 1997; Pough et al., 1987**)**. Although Perry et al. (2009) found that prescribed burning on 2–5 year intervals only reduced leaf litter amounts by 3% relative to unburned stands in the Ouachita Mountains of Arkansas, Matthews et al. (2009) found almost 4-fold decreases in leaf litter following two burns in the southern Appalachians. Immediate reduction in forest floor cover may have been the sole impact to woodland salamanders in our study as reductions in mid-story woody vegetation may actually reduce overall stand evapo-transpiration that in turn retains micro-site moisture benefiting woodland salamanders somewhat as well as increasing light to stimulate the response of forest floor vegetation (Homyack and Haas, 2008; Love et al., 2007).

That we observed fewer salamanders of all three species under coverboards in plots protected from deer herbivory also supports our hypothesis that woodland salamanders did respond at least behaviorally to changes in forest floor structure and composition. Protection from deer herbivory in burned plots allowed the prolific sprouts of wild grape (*Vitis* spp.) and other seedlings and forbs following the first fire (though killed following the second fire) and allowed considerable blackberry establishment following the second burn. Similar to responses in southeastern Ohio following burning in hardwood stands (Hutchinson and Sutherland, 2000), this quickly provided a sheltering forest floor cover that ameliorated leaf litter losses attributable to burning or reduced litter inputs from mid-story mortality on our study area. In shelterwood harvests in southern Ontario, Morneault et al. (2004) linked the rapid recovery of red-backed salamanders to the equally quick recovery of understory vegetation.

Our inability to detect great disparities in the ratio of adult to juvenile mountain dusky and red-backed salamanders among treatments suggests that conditions were not so degraded as to limit reproduction, create sink habitat conditions (Welsh et al., 2008), or cause noticeable competitive displacement of juveniles from coverboards (Moore et al., 2001) despite having natural cover component limited by reduced leaf litter. Marsh and Goicochea (2003) cautioned that coverboard sampling in the central Appalachians was biased (but see Houze and Chandler, 2002) towards adults with juveniles and smaller individuals proportionally using natural cover such as leaf litter to a greater extent. Using less-biased time-constrained searches methodologies, Homyack and Haas (2008) were unable to observe consistent patterns in adult to juvenile salamander ratios or over the wide array of harvest systems over 13 years.

Our *a priori* contrasts suggested that the larger-bodied slimy salamander with a lower surface area to weight ratio and a better ability to maintain hydration in drier micro-habitats (Grover and Wilbur, 2002) was even less affected in the years with fires or subsequent years than the other two common species in our study. Although we observed slimy salamanders in all months, their numbers were greatest during mid-summer when mountain dusky and red-backed salamander above-ground activity was reduced relative to the wetter spring and early summer months or cooler autumn months (Moore et al., 2001). Changes in micro-habitat conditions from prescribed burning as a disturbance process in this system may provide slimy salamanders a competitive advantage (Adams, 2007). This mechanism merits further investigation. The relatively simple salamander assemblage on our study site largely was composed of three very common, widespread species that have shown considerable resistance to disturbance (Ford et al., 2002a,b; Gibbs, 1998; Messere and Ducey, 1998). Because of this, Riedel et al. (2008) suggested that use of species such as the red-backed salamander as bio-sentinels of indicators of ecosystem integrity should be reevaluated when making natural resource management decisions that might affect numerous system components. As such, our results might not be as applicable in the areas of the central or southern Appalachians where a more diverse woodland salamander community occurs or where rare or sensitive species are present.

5. Management implications

For managers in the central and southern Appalachians, the ability to apply two prescribed fires in close temporal proximity without causing biologically significant harm to woodland salamanders should be seen as a positive outcome. However, much more remains to be learned about reintroducing fire as a regularly used forest management tool in the region. Fire seemingly has benefits for Anurans in general (Russell et al., 1999; Philliod et al., 2003; Hossack et al., 2009), specifically toads (*Bufo* spp.) in the central and southern Appalachians (Greenberg and Waldrop, 2008; Kirkland et al., 1996). Responses by Ambystomid salamanders or *Desmognathus* species more closely tied to the forest-riparian interface (i.e., dusky salamander, *Desmognathus fuscus*, or seal salamander, *D. monticola*, in the central and southern Appalachians) have not yet been examined. Also, we lack understanding about the relationship between woodland salamanders and their invertebrate prey (Harper and Guynn, 1999). Rieske-Kinney (2006) suggested that burning in hardwood forests decimates Collembola (springtails) numbers, an important diet item for most woodland salamander species (Petranka, 1998). Conversely, Love et al. (2007) failed to find negative impact from a one-time prescribed fire to macroinvertebrate abundance or community composition that could be correlated to woodland salamanders in the southern Appalachians. Yet, these authors did not speculate if this relationship would be maintained over successive fires. Lastly, because timber harvest does have an impact on woodland salamanders in the central Appalachians for at least a few years post-harvest (Homyack and Haas, 2009), we believe that any significant interaction between harvesting and fire might occur not with preparatory burns as in our study, but rather in fires that might be applied immediately following timber harvest (Brose et al., 1999).

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