



Effects of timber harvest within streamside management zones on salamander populations in ephemeral streams of southeastern Kentucky



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ABSTRACT

Timber harvest is an important extractive, economic activity to many human economies, but it can be detrimental to ecosystem function and species viability therein by degrading and fragmenting forest habitat. Salamanders comprise a significant amount of forest community biomass, and given their sensitivity to environmental stressors, including those caused by timber harvest, they often serve as important indicators of declines in forest ecosystem function. Several studies have focused on the impacts of timber harvest on salamanders inhabiting perennial and intermittent streams, the findings of which have helped inform best management practices for timber harvest in the U.S. Ephemeral headwater streams and associated riparia account for a small fraction of the total landscape, yet these features are critical to the functioning of forested ecosystems; however, few studies have examined how timber harvest impacts salamanders in or near these areas. Our objective was to investigate the effects of three different silvicultural treatments, each involving different streamside management zone (SMZ) characteristics, on salamander communities in southeastern Kentucky hardwood forest ephemeral streams. Data were collected by regular checks of pitfall traps, coverboards, and transect searches. Using both pre- and post-harvest data, abundance estimates were acquired using binomial mixture models. Declines in some species of terrestrial and stream-breeding salamanders were detected, and were shown to be likely related to characteristics of the corresponding silvicultural treatment. We suggest that application of modest SMZ regulations to ephemeral streams would likely reduce or alleviate salamander declines in these important headwater areas.

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1. Introduction

Headwater streams and associated riparia account for a small fraction of the total landscape, yet these habitats are critical to the functioning of forested ecosystems. These areas are involved in regulation of soil moisture, preserving nutrients and soil from runoff and erosion, and influencing air, water, and soil temperatures (Lowe and Likens, 2005). However, headwater streams and

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riparia are sensitive to damage from anthropogenic changes to forested landscapes, particularly changes associated with timber harvest (Brown et al., 1997).

Numerous federal, state, and local regulations have been implemented to protect streams from timber harvest, including Streamside Management Zones (SMZs), otherwise referred to as stream buffers. SMZs typically have requirements for improved crossings, road construction, and the preservation of standing timber, dictated by stream characteristics including classification into perennial, intermittent, or ephemeral types. Perennial streams are often afforded the most liberal protections; in Kentucky, this includes leaving an unharvested stream buffer free of trails, roads, and landings that range in width from 7.6 m to 50.3 m depending on bank slope (Stringer and Perkins, 2001). Additionally, 50% of canopy trees must be preserved within 16.8 m on either side of the perennial streams with bank slopes >15%; no canopy retention

requirements exist for intermittent streams; however, a 7.6 m buffer is required on flat ground, and the buffer increases in width by 1.5 m for every 5% increase in bank slope (Stringer and Perkins, 2001). Conversely, no buffer or canopy retention requirements exist for ephemeral streams in Kentucky, and few exist anywhere in the eastern United States (Witt et al., 2013).

Salamanders (Plethodontidae) are the dominant vertebrate in low order streams and riparia within eastern North America, and can substantially contribute to the biomass of these environments (Peterman et al., 2008). Numerous studies have demonstrated that salamander populations are particularly vulnerable to large scale anthropogenic landscape disturbances (Semlitsch et al., 2009; Price et al., 2011). Specifically, timber harvests can be particularly detrimental to many terrestrial breeding salamander species, populations of which may require long periods of time for full recovery (Petranka et al., 1993; Connette and Semlitsch, 2013). Many terrestrial species are dependent on key microhabitat variables such as surface moisture and canopy cover (Peterman and Semlitsch, 2013), which can be affected by timber harvest (Petranka et al., 1993). In addition, changes to the in-stream habitat of low-order streams after timber harvests can cause decreased abundances of stream salamanders; these declines have been shown to likely result from logging-associated sediment inputs (Lowe and Bolger, 2002; Lowe et al., 2004; Moseley et al., 2008; Peterman and Semlitsch, 2009). For example, Peterman and Semlitsch (2009) found that sediment associated with even-aged timber harvest was the only habitat variable they measured that was negatively associated with larval two-lined salamander (*Eurycea wilderae*) abundance.

Numerous studies have looked at plethodontid salamander populations at sites with different histories of timber harvest (Petranka et al., 1993; Ford et al., 2002; Lowe and Bolger, 2002; Knapp et al., 2003; Crawford and Semlitsch, 2008; Moseley et al., 2008; Peterman and Semlitsch, 2009). However, the use of before-after control-impacted (BACI) studies to evaluate the response of salamander populations to timber harvest are uncommon (but see Perkins and Hunter, 2006). BACI studies are often preferred to control versus impacted designs because they incorporate both time and control sites and they can alleviate the chance that variation in unmeasured covariates among sites are influencing observed effects (McDonald et al., 2000). Because salamander populations can be distributed unevenly spatially and temporally (Wyman, 1988; Connette and Semlitsch, 2013; Peterman and Semlitsch, 2013), assuming pre-treatment site homogeneity can potentially weaken experimental conclusions (deMaynadier and Hunter, 1995).

We conducted a BACI study to examine salamander populations in a managed, mixed-mesophytic forest of southeastern Kentucky. Specifically, we examined how timber harvest using the current Kentucky Best Management Practices (BMPs) affected salamander abundances in ephemeral streams and the adjacent riparian habitat if SMZ regulations similar to those for intermittent streams were applied to ephemeral streams.

2. Methods

2.1. Study site

Our study was conducted in the main block of the University of Kentucky's Robinson Forest (RF), located in Breathitt and Knott counties, in southeastern Kentucky. The main block of RF contains 4450 ha of relatively intact second growth deciduous forest. Elevations range from approximately 243–487 m (Overstreet, 1984). All roads are dirt or gravel, and most stream crossings are unimproved. The predominant forest assemblage is characterized as

mixed mesophytic, including roughly 30 co-dominant tree species (Braun, 1950). Common tree species include American beech (*Fagus grandifolia*), yellow-poplar (*Liriodendron tulipifera*), basswood (*Tilia spp.*), sugar maple (*Acer saccharum*), northern red oak (*Quercus rubra*), white oak (*Quercus alba*), eastern hemlock (*Tsuga canadensis*), and yellow buckeye (*Aesculus octandra*) (Braun, 1950). Understory species included eastern redbud (*Cercis canadensis*), flowering dogwood (*Cornus florida*), spicebush (*Lindera benzoin*), pawpaw (*Asimina triloba*), umbrella magnolia (*Magnolia tripetala*), and bigleaf magnolia (*Magnolia macrophylla*). Ridge tops, south facing slopes and areas with rocky shallow soils are characterized by oak-hickory (*Quercus-Carya*) and oak-pine (*Quercus-Pinus*) communities (Overstreet, 1984).

Both pre- and post-harvest salamander sampling was conducted in 11 randomly selected ephemeral streams in 6 watersheds, all within the 1545 ha Clemons Fork drainage. Our study sites were selected at random from a pool containing all the ephemeral channels in both watersheds of each treatment. We defined ephemeral streams as those which flow only during short periods of surface runoff events, such as after snowmelt or heavy rainfall (Fritz et al., 2008). The watersheds ranged from 25–60 ha, were located in the same elevation range (305–378 m), and all had bank slopes exceeding 15% (Schneider, 2010).

2.2. Timber harvest methods

Between June 2008 and March 2009, four first-order watersheds were harvested. A two-age deferment harvest (shelterwood with reserves system) was applied, resulting in a two-age stand with a residual target basal area of 3.4 m² per ha of reserve trees (4 dominant or co-dominant trees per ha) (Witt, 2012). This method was used over the entirety of the watersheds, with the exception of landings, trails, and the areas subject to SMZ treatments. Blocking of ephemeral channels with logging debris was not permitted, in accordance with Kentucky's BMP regulations (Stringer and Perkins, 2001).

The ephemeral streams included in this study were subjected to one of three treatments. Treatment 1 ($n = 3$) was designed to reflect the current SMZ requirements for ephemeral streams (no buffers or basal area retention). Additionally, no improved crossings were used for ephemeral streams assigned to treatment 1. Machinery crossed the streams at right angles, and material moved during skid trail construction was placed in areas not susceptible to erosion into ephemeral channels (Witt, 2012). Treatment 2 ($n = 4$) consisted of guidelines similar to those currently applied to intermittent streams including a 7.6 m buffer and the retention of a tree stringer (defined as retaining the canopy tree nearest to the stream bank along the length of the channel). Additionally, improved crossings were used for streams assigned to treatment 2. Crossings were composed of wooden skidder bridges, steel culverts, or PVC pipe bundles (Mason and Moll, 1995). Typically, skid trail stream crossings were in use for a two to six week period, and were removed after the area was harvested. The third treatment ($n = 4$) consisted of a no-harvest control.

All skid trails were constructed with a bulldozer, typically along the contour intervals. The most common vehicles using stream crossings included rubber tired cable or grapple skidders, although occasional crossings were made by tracked machines such as feller bunchers and bulldozers (Witt, 2012). After the harvests were completed, skid trails were retired in accordance with Kentucky's BMP law (Stringer and Perkins, 2001). This entailed the removal of all improved crossing structures, building of permanent water control structures ("water bars"), and seeding of the skid trail surfaces adjacent to ephemeral stream channels.

2.3. Salamander sampling methods

Salamanders were sampled using a combination of drift fences and pitfall traps, and visual encounter surveys that included coverboards, leaf litter searches, and stream transect searches both before and after timber harvest. All harvests were conducted during daylight hours. For the pre-harvest study, the sites were sampled once per month from 9 March 2007 to 21 November 2007 (except during May 2007), and from 14 March to 17 May 2008. After the harvests, sampling was conducted once per month from 23 September to 24 November 2011, and from 27 March to 4 November 2012. Each sampling period consisted of 3–14 days, timed with a rainfall event if possible, during which pitfall traps were opened continuously and checked daily. Cover board checks, leaf litter searches, and stream transects were performed once per sampling period.

One drift fence array was present at each site, and the pitfall trap was located on a randomly chosen side of the stream channel. The pitfall trap consisted of four 13.3L buckets buried flush with the ground, one in the middle and three spread out at 120 degree angles (“Y”-shaped) at the end of a 15.2 m drift fence. One arm of the drift fence ended at a perpendicular intersection with the stream channel. The drift fences consisted of erosion control fence (landscape fabric) buried into the soil and stapled over stakes for support, and were typically 40 cm high.

Twenty coverboards were placed at each site, ten on each side of the channel and all within 5 m of the stream edge. Cover boards were originally placed in 2005, and were replaced when any measurable piece broke off (Schneider, 2010). The boards were composed of 60 cm × 60 cm sheets of untreated plywood, 1.5 cm thick, similar to those used in other studies (Houze and Chandler, 2002; Marsh and Goicochea, 2003).

Stream transect searches were conducted along either a 50 m or 5 m transect. During stream sampling, every moveable cover object larger than 10 cm in diameter was overturned, and any salamanders revealed were captured or at least identified to genus level. For pre-harvest sampling and for post-harvest sampling periods from 23 September 2011 thru 24 November 2011, only a 5 m stream transect was conducted, but from March 2012 onwards, a 50 m transect was used. The location of the 5 m transect was determined randomly, whereas the 50 m transect was conducted with 25 m above and below the point at which the drift fence arm adjoined the stream channel. For comparisons using both pre- and post-harvest data, we randomly reduced the 50 m transect capture dataset to 5 m to enhance comparability among transect lengths. Residual basal area was also measured post-harvest using prism plots at 10 m intervals along the 50 m transect. These measurements were averaged to produce one residual basal area value for each sampling site.

Ten leaf litter searches were conducted once per sampling round at each site, five at random locations along either side of the stream within 5 m of the channel. In accordance with Schneider (2010), each litter search involved a 50 cm² patch of ground within 5 m of the stream bank which was removed of leaf debris one piece at a time until bare dirt was exposed. Any salamanders uncovered were captured, or at least identified to genus level.

Meteorological data, including rainfall and daily minimum temperature, were obtained from a permanent weather station (data loggers used were Campbell Scientific CR10X, Campbell Scientific, Logan, Utah, USA) located on the northwest bank of Clemons Fork about three kilometers south of the intersection of Clemons Fork and Little Millseat Branch and an average of 3.01 km from our sampling sites.

2.4. Estimating salamander abundance

We used a binomial mixture model to analyze treatment effects (Royle, 2004). This model uses spatially and temporally replicated

counts to estimate abundance, survey and site-level covariates, and detection probabilities, while providing estimates of uncertainty affiliated with each parameter (Dodd and Dorazio, 2004; Price et al., 2011). The model is defined as:

$$\prod_{j=1}^J [y_j | N_j, p_j] \prod_{i=1}^I \text{Bin}(y_{ij} | N_i, p_i) \quad (1)$$

where N_i represents animals sampled, p_i represents detection probability, and J represents sampling events.

We treated abundance estimates of four different species (*Eurycea cirrigera*, *Notophthalmus viridescens*, *Plethodon glutinosus*, and *Plethodon richmondi*) and one genus (*Desmognathus*) as individual response variables. We assumed salamander abundance may differ between treatments and basal area, and therefore considered the site-level abundance to be modeled with Poisson distribution.

We modeled our individual detection probability (p) following a binomial distribution (Kéry et al., 2009), and the model is defined as:

$$Y_{ij} | N_i \sim \text{Bin}(N_i, p_{ij}) \quad (2)$$

To account for differences in salamander activity and detectability between sites and sampling rounds, we used temperature (°C) and rainfall (cm) data corresponding to the time of sampling as covariates. We modeled heterogeneity based on these variables as:

$$Y_{ij} | N_i \sim \text{Bin}(N_i, p_{ij}) \\ \text{logit}(p_{ij}) = \alpha_0 + \alpha_1 \cdot \text{temperature} + \alpha_2 \cdot \text{rainfall} \quad (3)$$

After collection from the weather station, temperature and rainfall data were standardized to a z-score with a mean of 0 and a standard deviation of 1 before being incorporated as covariates in the model.

We used WinBugs version 1.4.3 (Spiegelhalter et al., 2003) to estimate population parameters, and relied on R version 2.15.2 (Venables and Smith, 2012) for additional data analysis. We used non-informative priors, and in accordance with Royle and Dorazio (2008), assumed $\beta \sim N(0, 10^2)$, $\alpha_0 \sim N(0, 1.6^2)$, $\alpha_1 \sim N(0, 10^2)$, $\alpha_2 \sim N(0, 10^2)$. Posterior summaries for parameters were based on three chains (Markov chain Monte Carlo) with 100,000 iterations, with a 20,000 sample burn in and a thinning rate of 5. We used the Gelman–Rubin r-hat statistic to ensure convergence. Abundance estimates were log transformed via $[(\exp(\beta_0) \cdot \exp(\beta_1 \cdot \text{treatment}))(\beta_2 \cdot \text{basal area})]$. Means and standard deviations for each model coefficient were calculated, along with 2.5 and 97.5 distribution percentiles, representing 95% Bayesian credible intervals.

3. Results

Pre-harvest sampling conducted from 9 March 2007 to 17 May 2008 resulted in 408 salamander captures, and post-harvest sampling conducted from 23 September 2011 until 4 November 2012 resulted in 382 salamander captures (Table 1). Salamanders from both collection periods included 11 species. After reduction of our dataset to conform to pre-harvest transect methodology, this resulted in a post-harvest total of 233 salamander captures.

Temperature and rainfall were significantly associated with detection of salamanders in both pre- and post-harvest sampling; this was true for a negative association between *P. richmondi* and temperature and a positive association between *Desmognathus* and rainfall. (Table 2).

We used both pre- versus post-harvest comparisons and control versus treatment comparisons (Table 3). The model revealed no significant differences in salamander populations between any of

Table 1
Salamander captures organized by species and method.

Species	Method				Total
	Drift fence/ pitfall	Coverboard	Stream transect	Leaf litter	
<i>Ambystoma maculatum</i>	3	0	0	0	3
<i>Desmognathus</i> spp.	6	4	60	0	70
<i>Eurycea bislineata</i>	12	20	7	1	40
<i>Eurycea longicauda</i>	0	1	0	0	1
<i>Gyrinophilus porphyriticus</i>	3	0	0	0	3
<i>Hemidactylum scutatum</i>	0	0	0	1	1
<i>Notophthalmus viridescens</i>	86	18	5	1	110
<i>Plethodon glutinosus</i>	8	261	1	7	277
<i>Plethodon richmondi</i>	19	64	2	25	109
<i>Pseudotriton ruber</i>	27	1	0	1	29
Totals	164	369	75	35	643

Table 2
Covariate means and 95% credible intervals, organized by genera and species for pre-harvest salamander sampling conducted from 9 March 2007 to 17 May 2008, and for post-harvest sampling conducted from 23 September 2011 until 4 November 2012. Post-harvest results are italicized.

Species	Temperature	Rainfall
<i>Desmognathus</i> spp.	-0.051 (-0.43, 0.31) <i>-0.0072 (-0.16, 0.18)</i>	1.60 (0.66, 2.60) ^a <i>0.282 (0.07, 0.49)^a</i>
<i>Eurycea cirrigera</i>	0.11 (-0.31, 0.51) <i>-0.32 (-0.82, 0.17)</i>	-0.25 (-1.41, 0.85) <i>-1.49 (-2.97, -0.224)^a</i>
<i>Notophthalmus viridescens</i>	0.38 (0.12, 0.64) ^a <i>0.10 (-0.25, 0.47)</i>	0.015 (-0.71, 0.72) <i>-0.013 (-0.71, 0.66)</i>
<i>Plethodon glutinosus</i>	0.16 (0.0020, 0.32) ^a <i>-0.058 (-0.30, 0.18)</i>	-0.13 (-0.57, 0.30) <i>-0.33 (-0.84, 0.17)</i>
<i>Plethodon richmondi</i>	-1.50 (-2.00, -1.11) ^a <i>-1.04 (-1.70, -0.45)^a</i>	1.12 (0.59, 1.68) ^a <i>0.026 (-1.26, 1.22)</i>

^a Denotes significant association with detectability.

the watersheds before harvests were conducted. However, when different treatments were compared before and after timber harvests, ephemeral streams that received no SMZ buffers (treatment 1) had significantly fewer *Plethodon glutinosus* than we observed during pre-harvest sampling. When harvested watersheds were compared to unharvested based on solely post-harvest data, the unharvested had significantly higher abundances of *Desmognathus* spp. salamanders in addition to higher abundances of *Plethodon glutinosus* (Table 4). Despite the lack of significance for treatment effects on other species, based on small amounts of overlap in some of our Bayesian credible intervals it is likely from the results that fewer salamanders were captured in ephemeral streams that received no SMZ buffers (treatment 1). Ephemeral streams that were prescribed regulations similar to the Kentucky BMP SMZ regulations for intermittent streams (treatment 2: 7.6 m buffer, improved crossings, and retention of a tree stringer) did not differ in salamander abundance compared to control sites in either comparison. Additionally, the covariate residual basal area was positively associated with *Plethodon glutinosus* abundance (Table 4).

As a variable, the control treatment was associated with higher abundance of *Plethodon glutinosus* in both pre- and post-harvest sampling. Treatment 1 status was associated with lower abundances of *Eurycea cirrigera* in post-harvest sampling only. Treatment 2 was associated with higher abundances of all salamanders except for *E. cirrigera* in pre-harvest sampling, and

Table 3

Binomial mixture model abundance estimate means and 95% Bayesian credible intervals, organized by genera and species, for pre-harvest salamander sampling conducted from 9 March 2007 to 17 May 2008, and for post-harvest sampling conducted from 23 September 2011 until 4 November 2012. Post-harvest results are italicized.

Species	Control (n = 4)	Treatment 1 (n = 3)	Treatment 2 (n = 4)
<i>Desmognathus</i> spp.	4.80 (1.42, 13.92) <i>4.75 (1.88, 10.53)</i>	3.25 (0.56, 10.65) <i>1.15 (0.14, 3.56)</i>	4.03 (1.10, 11.70) <i>2.48 (0.70, 6.21)</i>
<i>Eurycea cirrigera</i>	1.93 (0.30, 6.40) <i>4.88 (1.09, 17.64)</i>	2.31 (0.40, 7.26) <i>0.81 (0.018, 3.74)</i>	4.51 (0.99, 14.91) <i>5.59 (1.16, 20.96)</i>
<i>Notophthalmus viridescens</i>	4.53 (1.40, 13.22) <i>6.75 (1.09, 27.55)</i>	6.14 (2.00, 17.72) <i>14.95 (2.88, 60.38)</i>	9.33 (3.20, 28.21) <i>15.19 (3.43, 58.68)</i>
<i>Plethodon glutinosus</i>	17.21 (9.97, 28.98) <i>9.59 (4.79, 18.27)</i>	9.36 (4.66, 16.82) ^a <i>1.37 (0.27, 3.43)</i>	9.10 (4.71, 16.13) <i>3.91 (1.54, 7.94)</i>
<i>Plethodon richmondi</i>	24.49 (5.75, 97.26) <i>3.49 (0.78, 10.92)</i>	15.64 (3.12, 62.93) <i>1.25 (0.13, 4.23)</i>	20.46 (4.43, 84.24) <i>2.70 (0.55, 8.54)</i>

^a Denotes significant difference between pre- and post-harvest abundance.

with all salamanders except for *Desmognathus* spp. and *Plethodon richmondi* in post-harvest sampling.

4. Discussion

Our results show that when ephemeral streams are provided protection similar to that of Kentucky SMZ regulations for intermittent streams, declines in salamander abundances can be mitigated. Ephemeral streams with buffers and canopy retention did not differ significantly from our control sites in any of the analyses. Salamander abundances in ephemeral streams that lacked SMZ buffer protection differed on several occasions compared to pre-treatment data or controls. Abundances for *Desmognathus* spp. and *P. glutinosus* in unprotected streams were lower than those for control streams. Additionally, abundances for *P. glutinosus* were higher during pre-treatment compared to post-treatment in streams where no SMZ buffers were used. In order to maintain ecosystem function in riparian forest habitats, adding an SMZ requirement for ephemeral streams is warranted. Ephemeral stream SMZs should include, at a minimum, the preservation of the overstory tree nearest to the stream bank. Other research has shown the benefits of increasing the buffer width to a greater extent; this may also be warranted (Semlitsch and Bodie, 2003); for example, Peterman and Semlitsch (2009) found that a stream with a 30 m buffer did not differ from an unharvested control stream in terms of impacts on larval salamanders. Prescribing the precise width of SMZ buffers may be dependent on local features such as bank slope and soil type. Therefore, managers might consider developing ephemeral stream SMZ guidelines for their region or state that provide a range of buffer width values rather than one universal value.

It has been indicated that sedimentation is a likely pathway through which stream salamander abundances decline as a result of some timber harvests (Corn and Bury, 1989; Lowe and Bolger, 2002; Peterman and Semlitsch, 2009). While we did not measure sedimentation directly, other research conducted on our study sites and during the same time period

Table 4

Binomial mixture model abundance estimate means and 95% Bayesian credible intervals, organized by genera and species, for sampling conducted from 23 September 2011 until 4 November 2012. These abundance estimates derive from a model including residual basal area as a continuous covariate, each treatment value is an average of the sites within it. Covariate means and 95% confidence intervals for residual basal area are also shown.

Species	Control (n = 4)	Treatment 1 (n = 3)	Treatment 2 (n = 4)	Covariate value
<i>Desmognathus</i> spp.	12.85 (7.27, 20.82)	2.59 (0.93, 5.24)	7.48 (3.84, 12.67)	0.0091 (−5.77E ^{−5} , 0.018)
<i>Eurycea</i> <i>bislineata</i>	4.20 (0.92, 13.91)	0.68 (0.017, 2.85)	4.75 (0.93, 16.01)	0.0044 (−0.014, 0.022)
<i>Notophthalmus</i> <i>viridescens</i>	5.77 (0.92, 20.02)	12.90 (2.55, 44.93)	13.53 (2.72, 49.99)	−0.0067 (−0.023, 0.0085)
<i>Plethodon</i> <i>glutinosus</i>	11.77 (4.94, 25.08)	1.60 (0.30, 4.22) ^a	4.86 (1.13, 8.65)	0.016 (0.0047, 0.028)
<i>Plethodon</i> <i>richmondi</i>	4.96 (0.75, 18.75)	1.56 (0.14, 5.85)	3.84 (0.53, 15.27)	0.021 (−1.62E ^{−4} , 0.042)

^a Denotes significant difference between pre- and post-harvest abundance.

indicated the importance of SMZs at reducing soil erosion when applied to ephemeral streams. Witt et al. (2013) studied ephemeral streams in the same watersheds used in our study and found that the ephemeral streams with buffers, improved crossings and retained tree stringers were not statistically different in terms of total suspended solids (TSS) than control streams. However, ephemeral streams that lacked buffers displayed TSS amounts 598% greater than control streams. Of the various types of improved crossing structures employed in our study area, wooden skidder bridges were better than other methods at reducing sedimentation, and were likely more cost-effective from the perspective of a logging company in that installation times were less, and multiple reuse was possible (Bowker, 2013). Therefore, we recommend the inclusion of improved crossings for ephemeral channels in new SMZ regulations, with skidder bridges being the preferred type.

The measured declines in a terrestrial species included in our analysis, *P. glutinosus*, are consistent with previous studies of terrestrial breeding salamanders (Knapp et al., 2003; Hocking et al., 2013). The declines may be a response to the degradation of key microhabitat variables often associated with timber harvest (Semlitsch et al., 2009; Peterman and Semlitsch, 2013), and therefore the preservation of such variables might reduce the decline in abundances of *P. glutinosus*. This is supported by the association between increased abundances of *P. glutinosus* with increased residual basal area, which follows other findings (Peterman and Semlitsch, 2013).

Differences in *P. richmondi* abundances were not detected between streams with and without buffers. We propose that this is due to differences in life history and physiology characteristics; increased fat storage in the tail compared to other species we studied and deep subterranean aestivation during the hottest months of the year might reduce their vulnerability to lower soil moisture and higher surface temperatures characteristic of a recently harvested forest (Green and Pauley, 1987; Petranka, 1998). This seems to be supported by the negative association we found between higher temperatures and *P. richmondi* captures based on both pre- and post-harvest data, which we did not find for *P. glutinosus* or *Notophthalmus viridescens*, the other terrestrial species we included in our analysis. However, our study used temperature and rainfall covariates based on data gathered during sampling, whereas weather patterns during the days previous to sampling may impact salamander activity as well.

Regionally, long term studies that examine salamander population responses to harvest treatments are largely lacking (but see Homyack and Haas, 2009), and it is unclear whether SMZs, current or enhanced, will have long-term benefits to salamander species associated with ephemeral streams. Because most of our sampling methods were biased towards adult salamanders, changes in the age structure of populations could have gone undetected in our study. Additionally, an unmeasured covariate such as timing of harvests or time since harvest could be affecting our results. Although our harvests were done simultaneously, differences in completion of harvests did vary within an eleven month window due to differences in watershed size.

Finally, the inclusion of supplementary habitat variables in future studies that employ a BACI design may help to guide additional recommendations by elucidating specific mechanisms responsible for observed changes in salamander abundances. Future studies should seek to determine these underlying causes of reductions in salamander abundances. It is also possible that salamander detection is affected by an aspect of timber harvest, which might give a false impression of declines. Increased data collection at night and increased attention to aspects of timber harvest that might reduce detection may account for this possibility in future studies.

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