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# Effective Use of Trails for Assessing Terrestrial Salamander Abundance and Detection: A Case Study at Great Smoky Mountains National Park

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**ABSTRACT:** The conservation and management of wildlife species is contingent on estimating distribution and abundance. Sampling of wildlife requires repeated visits to accurately determine species occurrence and to quantify abundance across temporal and spatial scales. The use of trails to sample wildlife populations is increasing and offers opportunities to potentially sample more frequently, with increased ease of access, and less disturbance to habitats, which can be important in sensitive natural areas. We examined capture data of terrestrial salamanders within Great Smoky Mountains National Park to determine if detection and abundance estimates from trail and non-trail transects were significantly different. Across two, 3-week periods during June and July 2012, we sampled 195 transects (70 along trails and 125 within non-trail habitat) on multiple occasions. We found that most microhabitat variables associated with salamander detection and abundance did not differ between trail and non-trail transects. Further, our models indicate detection and abundance of terrestrial salamanders were not significantly different on trail and non-trail transects. These results suggest trails can be used to accurately estimate abundance of terrestrial salamanders and may reduce the need to sample for plethodontid salamanders in sensitive habitat.

*Index terms:* *Desmognathus*, *Eurycea*, N-mixture, *Plethodon*, visual encounter surveys

## INTRODUCTION

Appropriate sampling techniques are essential to provide information for the conservation and management of species (Elzinga et al. 2001). In some instances, access to sites for sampling can be difficult when attempting to avoid areas with sensitive vegetation or animal species, dense vegetation, or geographical barriers. This can cause time delays or limit the number and spatial extent of sites visited within a given timeframe, which can potentially affect inferences. One method of sampling involves using existing trails to access remote locations. Wildlife responses to trails vary, and this could lead to differences in detection and/or abundance of animals on trail systems in natural areas. For example, various Neotropical mammals use trails in different ways, with some avoiding trails and others using trails exclusively, resulting in significant variation in detection probability across species (Harmsen et al. 2010). Several studies have investigated abundance, diversity, and ecology of bird species along trail systems (e.g., Miller et al. 1998; Whitney and Smith 1998; Sutter 2000). Miller et al. (1998) found that habitat near trails harbored more generalist species and were impacted by higher nest predation, and Sutter (2000) found that abundance of several bird species was more variable near trails. As such, trails can have both positive and negative effects on wildlife populations.

Edge effects in forests due to processes such as timber harvesting, road construc-

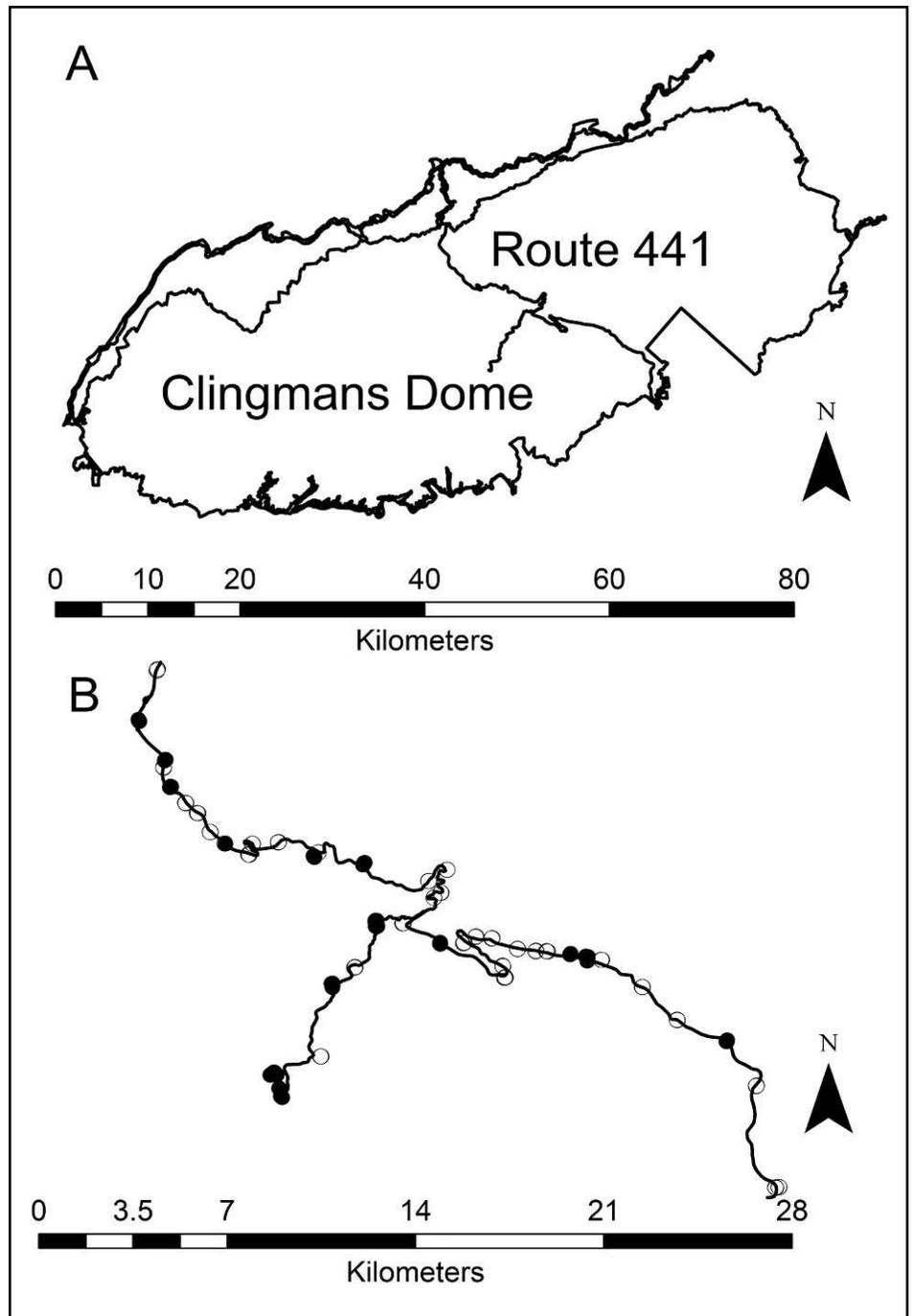
tion, and trail construction reduce patch sizes by creating boundaries with different (and generally less suitable) microclimates than interior forests (Murcia 1995; Harper et al. 2005; Hocking et al. 2013). The assumption when sampling for amphibians is that transects or plots should be established away from trails or trail systems, as these habitats could affect the abundance and richness of amphibian species (Cole and Landres 1995). The use of trails for sampling amphibians has not been thoroughly tested, but this technique has been used frequently to sample for amphibian disease, richness, and ecology in temperate and tropical regions (e.g., Duellman 1995; Smith et al. 2009; Barquero et al. 2010). Furthermore, two studies have found higher relative abundance of amphibians in habitats directly adjacent to trails compared to areas away from trails (Davis 2007; von May and Donnelly 2009). Although von May and Donnelly (2009) found higher relative density of frogs on trails in Australia, they cautioned that sampling on trails could bias estimates of abundance. Neither von May and Donnelly (2009), nor Davis (2007), specifically investigated the influence of trails on abundance of amphibians using methods to correct for spatial and temporal variation in detection probability.

Our study focused on using trails to conduct standardized surveys of plethodontid salamanders (Plethodontidae: *Plethodon*, *Desmognathus*, and *Eurycea*) to obtain unbiased estimates of abundance. Plethodontid salamanders are difficult to sample

due, in part, to their variable surface activity and small size. For example, Hyde and Simons (2001) used four different sampling techniques in Great Smoky Mountains National Park (GSMNP) to quantify plethodontid diversity and relative abundance, and found significant variability in salamander counts depending upon the sampling method used and habitat type being surveyed. Further, numerous studies have demonstrated that both individual and population-level detection probability of plethodontid salamanders is almost always less than one (Bailey et al. 2004a, 2004b, 2004d; Connette and Semlitsch 2013; Peterman and Semlitsch 2013). To calculate unbiased abundance estimates and to correctly infer how landscape covariates affect abundance, it is critical to use survey and statistical methods to account for imperfect detection (Royle 2004). The goal of this study was to determine whether conducting surveys for plethodontids using established recreational trails in Great Smoky Mountains National Park had a measurable influence on the individual detection probability and abundance of plethodontid salamanders.

## METHODS

We conducted nighttime visual encounter surveys (VES) to count surface-active plethodontid salamanders in terrestrial habitats within GSMNP, a natural area that straddles the border of North Carolina and Tennessee. We identified 70 potential survey sites based on accessibility from a road or trail that covered the elevational gradient from the lower limit of GSMNP to the top of Clingman’s Dome (highest point in GSMNP and third highest peak east of the Mississippi River) on both sides of the continental divide. For practical purposes, accessibility was defined as within 2 km of a parking lot or safe vehicle pull-off location and not separated from the road by any impassible barriers for researchers (e.g., cliffs, large rivers, etc.). We then randomly selected 15 high (1501–2025 m.a.s.l.), 15 mid (1001–1500 m), and 15 low (412–1000 m) elevation sites. The sites were distributed along US Route 441 and Clingman’s Dome Spur Road on both the North Carolina and Tennessee sides of the eastern continental divide (Figure 1).



**Figure 1.** (A) Boundary of the Great Smoky Mountains National Park, US Route 441, and the spur road to Clingmans Dome. (B) Locations of Trail (open circles) and non-trail (closed circle) transects along Route 441 and the spur road to Clingmans Dome.

Thirteen of these sites had recreational trails. We added three more sites that had trails from our initial pool of 70 to get a more even distribution of trail sites over the elevational gradient.

At each of the 48 selected sites, we established two to eight, 25 × 4-m transects (195 transects total). Transects located

along either trails (70 in total; hereafter trail) or a minimum distance of at least 20 m from a trail or road (125 in total; hereafter non-trail), were delineated by a labeled pin flag placed every 5 m along the 25-m transect. Sites had four transects each except in cases where accessibility prevented more than two, and except for sites with trails, where we established

four trail and four non-trail transects when possible. Edge effects of roads on terrestrial salamanders are minimized once a distance of 20 m from a road has been reached, and the effect within 20 m on terrestrial salamanders is suggested to be driven by impacts of vehicles (Marsh and Beckman 2004; Marsh 2007); therefore, we believe this distance was appropriate. Each site was visited on five occasions in June–July 2012 except one site that was visited opportunistically on one additional occasion during this time period (hereafter sample period). During each visit, transects were surveyed by one of four researchers walking a straight line following flags delineating 5-m markers searching 2 m to the left and right of the center line of the transect. For trail transects, this represented surveying the trail and approximately 1 m on each side of the trail, accounting for the existing width of the trail. Each transect was surveyed on 1–6 occasions (mean 4.3). Unequal survey efforts per transect were due to differences in the number of researchers present on a given night. For example, if there were four transects but only three researchers, only three transects were surveyed that night for logistical reasons. The statistical methods used to analyze these data are robust to unequal sampling efforts (described below).

Since surveys of transects were conducted periodically throughout the two month period, weather at the time of visitation was not taken into regard when sampling. Rather, survey-specific weather covariates were included in our hierarchical models as factors affecting salamander detection (see statistical analyses below). All salamanders encountered were visually identified to species largely in situ without handling; however, if a salamander was handled it was identified at the point of capture and immediately released. Paired transects running upslope were separated by approximately 20 m and surveyed simultaneously by researchers. The surveyed trails ranged from 0.5 to 3.0-m wide with most being approximately 1 m. These trails were maintained by the National Park Service and received high use from hikers, resulting in hard-packed soils, little intact leaf litter, and little, if any, vegetation in the center of the trail. They were typical of many trails

in the Appalachian Mountains in national parks or close to human population centers (D.J. Hocking, pers. obs., April 2014). We did not sample any trail sections within 5 m of bridges, stairs, ladders, or other man-made structures.

### Statistical Analysis

We used N-mixture models to account for imperfect detection and calculate abundances of Jordan's Salamanders (*Plethodon jordani* Blatchley), Pygmy Salamanders (*Desmognathus wrighti* King), and Blue Ridge Two-lined Salamanders (*Eurycea wilderae* Dunn; e.g., Royle 2004; Royle and Dorazio 2008). We built species-specific models that included site-level covariates to estimate abundance, including transect type (trail/non-trail), transect elevation, transect slope, northing, easting, topographic position (TPI), topographic wetness (TWI), percent canopy cover, percent herbaceous ground cover, leaf litter depth, and distance to nearest stream. A single, fully-parameterized model was fit for each species, and the importance of each covariate on abundance was inferred from this model. Topographic position represents a site's slope position relative to the surrounding landscape, and was calculated from a 10-m digital elevation model using a 100-m moving window (Dilts 2010). Topographic wetness was calculated accounting for solar insolation (azimuth = 180.0, elevation = 75.8; Theobald 2007). Canopy cover measurements were taken at three points using a densiometer. Percent herbaceous ground cover was visually estimated on a five-point scale where the value was determined as the closest value to one of the following categories: 0 = no cover, 1 = 25% cover, 2 = 50% cover, 3 = 75% cover, and 4 = 100% cover in a 1-m<sup>2</sup> area at three points in a transect. Leaf litter measurements were taken at three points along a transect using a ruler to the nearest mm; along trail transects, points were taken within 0.25 m adjacent to the edge of the trail (i.e., directly off trail). Each variable was measured at three locations along the centerline of each transect (0, 12.5, and 25 m), and the average of the three measures was used for statistical modeling. We recorded air temperature and relative humidity at each site during each

survey using a handheld weather meter (Kestrel Meters, Birmingham, MI). Lastly, we derived spatial rainfall maps describing the 24-hr cumulative precipitation across GSMNP. This was accomplished through spatial Kriging of rainfall estimates, based on temporal rainfall measures obtained from 24 weather stations located throughout GSMNP and the immediately surrounding area.

The covariates transect type, air temperature at the time of the survey, precipitation 24 hrs prior to survey, percent herbaceous ground cover, and relative humidity were included to estimate detection probability. We modeled abundance following a Poisson distribution with a log link to relate regression covariates to abundance (Royle 2004). We included a random site effect to account for potential autocorrelation among transects at a single site. We used a binomial distribution with a logit link to estimate detection covariate effects (Royle 2004). To improve model fit, we used an observation-level overdispersion term (i.e., random effect) in the detection sub-model (Kéry and Schaub 2012). We fit this hierarchical Bayesian N-mixture model in the program JAGS, implemented through R (R Core team 2013) using the rjags package (Plummer 2014). All continuous covariates were standardized to have a mean of 0 and standard deviation of 1, for model stability. In the Bayesian analysis, we used non-informative priors for all covariate effects following normal distributions with a mean of 0 and standard deviation of 10. We used a uniform prior between 0 and 10 for the random effect standard deviations (Gelman 2006). In statistics using Bayesian inference, the posterior predictive probabilities are a multiple of the likelihood and the prior probabilities of the model parameters. To make inference based on the data with minimal influence from prior information, we used these non-informative prior distributions that contain little distributional information. For example, with a uniform prior between 0 and 10, we are indicating that the standard deviation of the random site effect had an equal probability of being any value from 0 to 10 prior to adding information from the data. This results in virtually all of the inference being drawn

from the data (Gelman and Hill 2006). We used four Markov Chain Monte Carlo (MCMC) simulation chains with different random starting values and ran 400,000 iterations of each chain, discarded the first 300,000 as a burn-in phase, and thinned the remaining 100,000 by 40 to result in 10,000 iterations for statistical inference. We used independent *t*-tests to determine if site-level model covariates that significantly influenced abundance or detection of any species modeled (i.e., slope, leaf litter depth, and percent herbaceous ground cover) differed significantly between trail and non-trail transects. Daily weather variables were included in the detection process part of the N-mixture model; therefore, the estimates of abundance and the effects of trails are robust to variability in conditions while sampling.

## RESULTS

We observed 9522 salamanders of 14 species during our visual encounter surveys. From our N-mixture models, we estimated a mean abundance ( $\pm$ SD) of 44.5 ( $\pm$ 53.6) salamanders (of the three species modeled) across 125 non-trail transects, while we estimated a mean abundance of 34.8 ( $\pm$ 31.1) salamanders within the 70 trail transects. We estimated that mean abundances of 33.6 ( $\pm$ 41.5) *P. jordani*, 5.4 ( $\pm$ 10.6) *E. wilderae*, and 5.5 ( $\pm$ 11.1) *D. wrighti* were found on non-trail transects, and 26.7 ( $\pm$ 29.9) *P. jordani*, 4.6 ( $\pm$ 4.7) *E. wilderae*, and 3.5 ( $\pm$ 5.3) *D. wrighti* were found on trail transects. For non-trail transects, estimated densities ( $m^{-2}$ ;  $\pm$ SD) of *P. jordani*, *E. wilderae*, *D. wrighti*, and all salamanders combined were 0.34 ( $\pm$ 0.41), 0.05 ( $\pm$ 0.11), 0.05 ( $\pm$ 0.11), and 0.44 ( $\pm$ 0.53), respectively (Figure 2); while estimated densities on trail transects were 0.27 ( $\pm$ 0.30), 0.05 ( $\pm$ 0.05), 0.04 ( $\pm$ 0.05), and 0.35 ( $\pm$ 0.31) for *P. jordani*, *E. wilderae*, *D. wrighti*, and all salamanders combined (Figure 2). We found no significant effect of trail on the abundance of *P. jordani*, *E. wilderae*, or *D. wrighti* (Table 1).

The probability of detecting individuals that were on our transects was affected by temperature at the time of the survey, amount of precipitation in the 24 hours prior to the survey, percent herbaceous

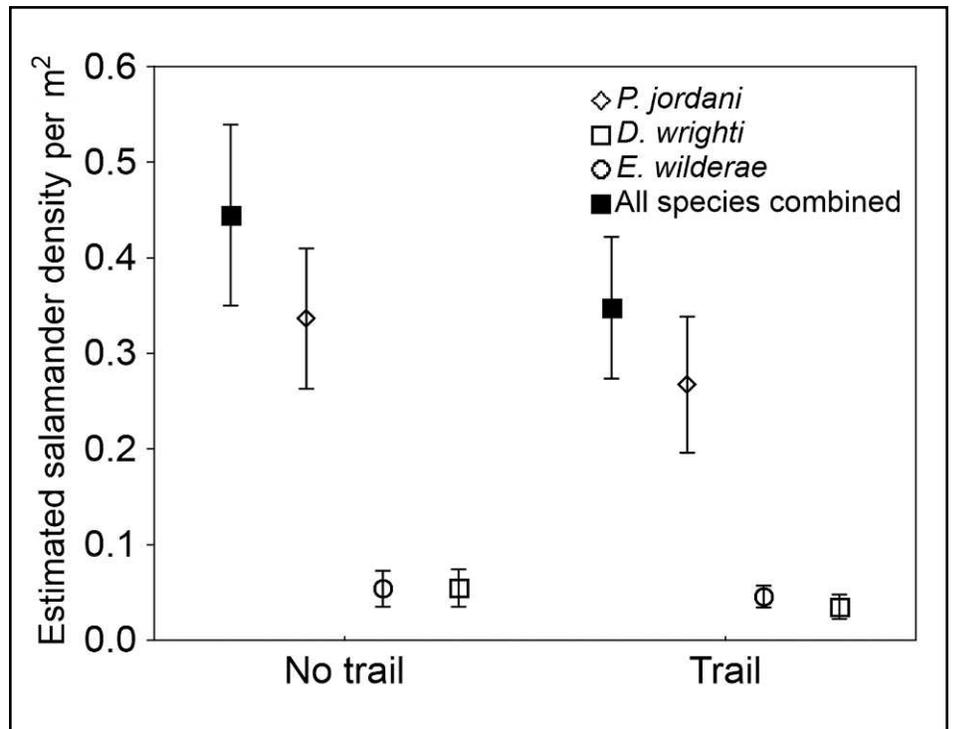


Figure 2. Mean estimated salamander density ( $\pm$ 95% CI; per  $m^2$ ) between trail and non-trail transects for *Eurycea wilderae*, *Plethodon jordani*, *Desmognathus wrighti*, and all species combined. Each site was located within Great Smoky Mountains National Park and was visited on five occasions in June–July 2012.

ground cover, and relative humidity (Table 1). Trail was included as a factor in the detection sub-model but was not significant for any of the species (Table 1).

Depth of leaf litter differed between trail and non-trail transects, with leaf litter depth being significantly greater in non-trail transects ( $t_{(1,193)} = 12.965$ ,  $P \leq 0.0001$ ). Ground cover ( $t_{(1,193)} = -0.302$ ,  $P = 0.763$ ) and slope ( $t_{(1,193)} = 1.067$ ,  $P = 0.287$ ) did not significantly differ between trail and non-trail transects. While there was a general trend for predicted abundance estimates to be slightly lower on trail transects, our models indicate detection and abundance of plethodontid salamanders did not differ significantly between trail and non-trail transects, but did differ as a consequence of slope, ground cover, leaf litter depth, and precipitation (Table 1).

## DISCUSSION AND CONCLUSIONS

Sampling on trails that traverse natural areas can be used effectively to quantify detection and abundance of terrestrial plethodontid salamanders. These results

could reduce the impact to undisturbed habitats, streamline sampling for terrestrial salamander species, and lead to a significant increase in the number of sites that can be visited during a sampling period. Terrestrial salamander detection and abundance were not significantly influenced by the location of transects (i.e., trails vs. non-trails), which indicates that vegetation and microclimatic differences across trail and non-trail transects did not significantly influence the probability of detecting salamanders, or the detection-corrected abundance of salamanders. These results corroborate several recent studies investigating the relative abundance of terrestrial salamanders within trail and non-trail habitats. For example, Fleming et al. (2011) and Davis (2007) found Red-backed Salamanders (*Plethodon cinereus* Green) at higher abundances near maintained trails in nature preserves in Ohio and Georgia, USA (respectively). Other herpetofauna, frogs and lizards, have also been found in higher relative abundance along trails in tropical rainforest (von May and Donnelly 2009). The studies above quantified relative abundance of terrestrial salamander

Table 1. The influence of environmental variables and trails (non-trail or trail transects) on abundance and detection of *P. jordani*, *E. wilderae*, and *D. wrighti* within GSMNP. Parameters presented without estimates were not included in the analyses for those species to ensure adequate mixing and convergence of the MCMC iterations. Mean coefficient estimates and 95% credible intervals were calculated from the hierarchical Bayesian regression model. Abundance coefficients are on log scale, and detection covariates are on the logit scale and were derived from standardized data so the relative effect of each parameter is comparable (e.g., Elevation has the largest effect on the abundance of all species, whereas the relative effects of detection parameters were more species specific).

Variable	<i>P. jordani</i>			<i>D. wrighti</i>			<i>E. wilderae</i>		
	Mean	2.5 % CI	97.5% CI	Mean	2.5 % CI	97.5% CI	Mean	2.5 % CI	97.5% CI
<b>Abundance</b>									
N-intercept	1.725	0.850	2.526	-1.647	-3.372	-0.164	0.490	-0.147	1.095
Elevation	1.985	1.235	2.948	1.978	0.605	3.569	3.495	0.491	6.476
Elevation <sup>2</sup>	-	-	-	-	-	-	-3.371	-6.447	-0.294
Slope	0.003	-0.475	0.499	1.653	0.094	3.285	-0.318	-0.674	0.041
Slope <sup>2</sup>	-0.299	-0.848	0.218	-1.158	-2.782	0.395	-	-	-
Northing	0.057	-0.056	0.170	0.284	-0.042	0.621	-	-	-
Easting	-0.029	-0.130	0.072	-0.218	-0.486	0.041	-	-	-
TPI	0.137	-0.005	0.279	-0.032	-0.439	0.359	-0.179	-0.509	0.143
Trail	0.059	-0.613	0.815	-0.105	-1.328	1.182	0.798	-0.092	1.810
Log(TWI)	0.059	-0.053	0.174	0.219	-0.032	0.468	-0.128	-0.372	0.115
Canopy	-0.024	-0.192	0.143	-0.130	-0.526	0.268	0.059	-0.388	0.546
Ground Cover	0.366	0.100	0.744	0.115	-0.459	0.611	-0.059	-0.396	0.303
Litter Depth	0.339	0.185	0.491	0.362	-0.025	0.755	0.045	-0.292	0.379
Log(Stream Dist)	-0.127	-0.330	0.082	0.177	-1.006	1.477	-0.443	-0.801	-0.101
Site Standard Deviation	2.291	1.694	3.133	3.199	2.115	4.720	1.289	0.933	1.754
<b>Detection</b>									
p-intercept	-1.062	-1.971	-0.317	-2.218	-3.083	-1.522	-1.825	-2.471	-1.245
Temperature	0.233	0.060	0.407	0.037	-0.751	0.784	0.221	-0.220	0.653
Temperature <sup>2</sup>	-0.152	-0.276	-0.041	-0.259	-0.679	0.136	-0.444	-0.769	-0.134
24-hr Precip	0.043	-0.036	0.127	0.389	0.121	0.680	0.463	0.222	0.717
Ground Cover	-0.498	-0.892	-0.193	0.639	-0.028	1.345	0.474	-0.040	0.959
Ground Cover <sup>2</sup>	0.134	0.017	0.251	-0.308	-0.660	0.036	-0.241	-0.537	0.048
Trail	-0.158	-0.877	0.479	0.291	-1.067	1.391	-0.090	-1.135	0.770
Rel. Humidity	0.133	0.025	0.244	0.903	0.475	1.382	0.723	0.374	1.101
Rnd. observation SD*	0.734	0.571	0.959	1.664	1.222	2.199	1.869	1.453	2.333

\* Standard deviation of the random observation effect

ders and other herpetofauna, and did not account for imperfect detection. Surface activity in plethodontid salamanders is highly variable (Bailey et al. 2004c) and may vary in relation to fine-scale habitat features (e.g., Peterman and Semlitsch 2013) or weather at the time of sampling (Connette and Semlitsch 2013; Peterman and Semlitsch 2013). Failure to account for imperfect detection can result in erroneous inference, where observation error occludes accurate estimate of the ecological process of interest (MacKenzie et al. 2006; Royle and Dorazio 2008). Our study found that trail and non-trail transects did not measurably influence our detection estimates of the GSMNP terrestrial salamander species included in this study (Table 1). Similarly, in a study investigating road edge effects, Marsh and Beckman (2004) found that *P. cinereus* detection did not differ between edge and interior habitats, and Northern Slimy Salamanders (*Plethodon glutinosus* Green) showed no significant decreases in abundance in edge habitat; however, edge did have an effect on *P. cinereus* abundance. The use of trails to accurately sample plethodontid salamander abundance could significantly ease the collection of data required to correct for imperfect detection when estimating abundance, while minimizing impacts to the environment.

Given the evidence from this study, trails appear to be an effective means to sample for nocturnally active terrestrial salamanders. Davis (2007) suggested an increase in microhabitat (e.g., coarse woody debris) was associated with terrestrial salamander (*Plethodon glutinosus*) presence on or near trails. In our study, although leaf litter depth (a microhabitat variable that significantly predicted the abundance of *P. jordani*) was significantly lower on transects located on trails (Figure 3), this did not measurably impact detection or abundance. There are likely two factors contributing to the lack of significant differences in estimated salamander abundance between trail and non-trail transects, despite significant habitat differences. First, although abundance of *P. jordani* was significantly affected by leaf litter depth, elevation had a much greater effect on abundance (Table 1). Second, while habitat measurements were collected immediately adjacent to the trails, home

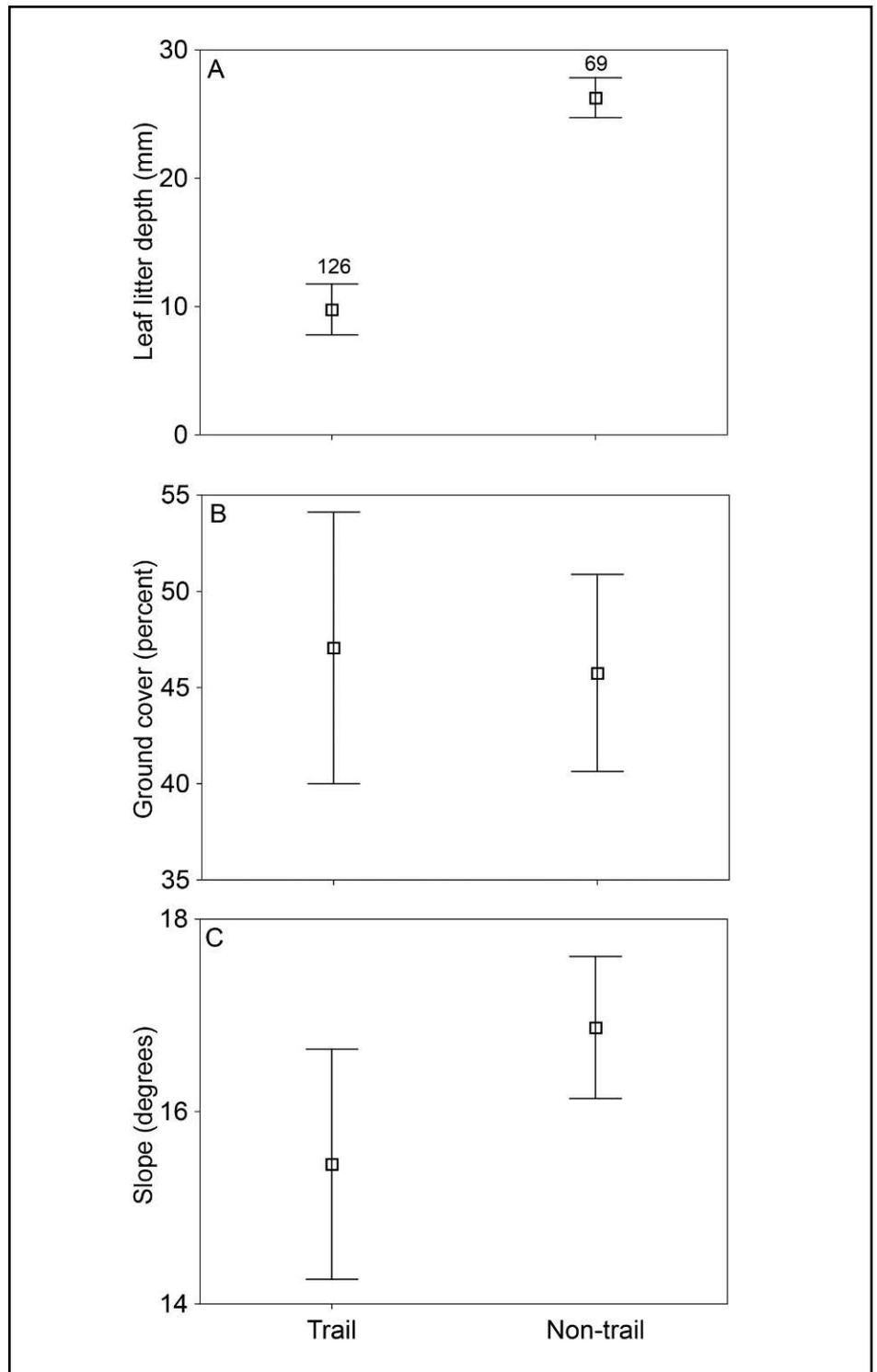


Figure 3. (A) Leaf litter, (B) percent herbaceous ground cover, and (C) slope across trail and non-trail transects. Whiskers represent standard errors ( $\pm 1$ ), and sample size for each variable is located above whiskers in A. Each site was located within Great Smoky Mountains National Park and was visited on five occasions in June–July 2012.

ranges of individual terrestrial salamanders extend beyond the trail into the surrounding habitat, thus individuals counted on trails likely experience microhabitats not

fully represented by our trail-side habitat sampling.

Another important aspect of our study is

our use of nighttime VES, which follows the guidelines of other studies (Hyde and Simons 2001). Sampling for plethodontids during diurnal hours could result in significant differences across plots with different amounts of leaf litter or surface cover objects. For example, Crawford and Semlitsch (2008) found that abundance of plethodontid salamanders was significantly lower in timber stands that had lower leaf litter depths and that diurnal area-constrained surveys for plethodontid salamanders resulted in significantly lower counts when compared to nighttime surveys. Peterman and Semlitsch (2013) found that detection of salamanders during daytime surveys was significantly affected by the amount of searchable cover. Our results suggest transect-level differences in microclimate and habitat altered detection and abundance of terrestrial salamanders at GSMNP (Table 1); however, if microclimate or habitat differences existed across trail or non-trail transects, they did not manifest to measurable differences in detection or abundance of terrestrial salamanders (Table 1).

Edge effects on species can be variable and dictated by the type of edge. Our study examined the effectiveness of trails as sampling tools, which are considered to have less impact than hard edges, such as edges from timber harvest cutblocks (edge between an open area and forest cover) or roads. For example, documentation of negative effects of roads and road edges on amphibians, reptiles, birds, and mammals is frequent and includes direct mortality, modification of behavior, increased nest predation, and reduced abundance (e.g., Trombulak and Frissell 2000; Fahrig and Rytwinski 2009; Butler et al. 2013). These effects are greatly minimized when edges are associated with smaller landscape-level disturbance, such as trails and gated roads. For example, Marsh (2007) found that gated roads had measurably less effect on *P. cinereus* abundance than ungated roads (roads not gated and open to vehicle travel). Additionally, terrestrial salamanders are also influenced by edge effects along heavily trafficked logging roads (deMaynadier and Hunter 2000). Our data suggest that trails that receive no vehicle traffic have little effect on terrestrial salamanders and, therefore, likely have minimal edge effects

as well.

The use of trails to conduct biological sampling could have greater implications with regard to sampling in sensitive habitats. The long-term effects of repeated visits to off-trail locations for sampling of wildlife or vegetation are not well documented; however, there is significant evidence of impacts from off-trail hiking on habitats, which is a similar activity to repeated sampling for biological reasons. For instance, measurable differences in plant height, stem length, and leaf area occur in areas trampled off-trail (see review in Cole 2004), which also reduces soil microbial community diversity (Zabinski and Gannon 1997) and vegetation cover (Boucher et al. 1991). Further, disturbance by off-trail visitors has been found to significantly alter behavior of wildlife in protected land in the United States (Taylor and Knight 2003; Stankovich 2008). These effects are compounded when sampling in high-use protected areas (e.g., national parks) where minimizing disturbance is a high priority due to the high number of visitors and greater proportion of sensitive habitat. Minimizing effects from biological sampling is also important for long-term monitoring efforts. In the face of increased climate and land cover change across the world, effects from repeated sampling could confound effects of other large-scale disturbances (e.g., global climate change). For example, Otto et al. (2013) showed that repeated sampling can decrease the detection probability of a terrestrial salamander (*P. cinereus*). Researchers are becoming increasingly aware of the importance of accounting for imperfect detection when estimating species occupancy or abundance. Because statistical models accounting for imperfect detection necessitate multiple visits to the same sites, use of survey methods that facilitate efficient sampling and that have minimal impact to the natural landscape should be preferred.

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