

Full Length Research Paper

Effects of microhabitat and large-scale land use on stream salamander occupancy in the coalfields of Central Appalachia

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Large-scale coal mining practices, particularly surface coal extraction and associated valley fills as well as residential wastewater discharge, are of ecological concern for aquatic systems in central Appalachia. Identifying and quantifying alterations to ecosystems along a gradient of spatial scales is a necessary first-step to aid in mitigation of negative consequences to aquatic biota. In central Appalachian headwater streams, apart from fish, salamanders are the most abundant vertebrate predator that provide a significant intermediate trophic role linking aquatic and terrestrial food webs. Stream salamander species are considered to be sensitive to aquatic stressors and environmental alterations, as past research has shown linkages among microhabitat parameters, large-scale land use such as urbanization and logging, and salamander abundances. However, there is little information examining these relationships between environmental conditions and salamander occupancy in the coalfields of central Appalachia. In the summer of 2013, 70 sites (sampled two to three times each) in the southwest Virginia coalfields were visited to collect salamanders and quantify stream and riparian microhabitat parameters. Using an information-theoretic framework, effects of microhabitat and large-scale land use on stream salamander occupancy were compared. The findings indicate that *Desmognathus* spp. occupancy rates are more correlated to microhabitat parameters such as canopy cover than to large-scale land uses. However, *Eurycea* spp. occupancy rates had a strong association with large-scale land uses, particularly recent mining and forest cover within the watershed. These findings suggest that protection of riparian habitats is an important consideration for maintaining aquatic systems in central Appalachia. If this is not possible, restoration riparian areas should follow guidelines using quick-growing tree species that are native to Appalachian riparian areas. These types of trees would rapidly establish a canopy cover, stabilize the soil, and impede invasive plant species which would, in turn, provide high-quality refuges for stream salamanders.

Key words: Canopy cover, coal mining, *Desmognathus*, *Eurycea*, headwater streams, surface mining.

INTRODUCTION

Amphibians, particularly salamanders, are an important component to both terrestrial and aquatic ecosystems

(Davic and Welsh 2004). Long-lived stream salamander species are a keystone faunal group, can reach high

densities in undisturbed areas, and their life history traits make them sensitive to both stream and upland watershed alterations (Welsh and Ollivier, 1998; Welsh and Droege, 2001; Southerland et al., 2004). In headwater streams, apart from fish, salamanders are the most abundant vertebrate predators. Stream salamanders provide a significant intermediate role in stream food webs, feeding on small prey such as benthic macroinvertebrates and are an important prey item for larger vertebrates (Petranka et al., 1993; Davic and Welsh, 2004; OHEPA, 2012). Undisturbed riparian areas of central Appalachia have relatively stable, intra- and inter-annual salamander populations (as compared to macroinvertebrates), with densities of up to 1.4/m² (Hairston and Wiley, 1993). Movement of many Appalachian stream salamander species has been shown to be less than 100 m (Pauley et al., 2000), and although adult stream salamanders are able to cross barriers, such as dry streambed sections and waterfalls, mobility may be limited by riparian and upper watershed habitat fragmentation (Grant et al., 2010; Resetarits 1997; Willson and Dorcas, 2003; OHEPA, 2012). Their low mobility reduces the likelihood of abundance being a reflection of immigration, which helps confirm linkages between local, long-term stream and watershed conditions and salamander survival and reproduction (Welsh and Ollivier, 1998; Welsh et al., 2005). Life history and behavior vary greatly among aquatic salamander species with some larval forms having aquatic phases of up to 48 months, whereas other species have very short- or even no aquatic larval phase (OHEPA, 2012). This biphasic life history trait shown by some salamander species can provide a gradient of responses to aquatic and upland environmental conditions giving better insight to both terrestrial and stream conditions (Fisher et al., 1998; Petranka, 1998). For example, some species of stream salamander abundances may be reduced, even to the point of local extirpation, by decreases in physiochemical stream and riparian conditions from watershed land uses such as mining, urbanization and timber harvesting that cause changes in pH, streambed sedimentation, suspended sediment, and water and soil temperature (Willson and Dorcas, 2003; Welsh et al., 2005; Moseley et al., 2008). Accordingly, other species of stream salamanders may be very tolerant of habitat degradation. Therefore, variance in species composition may be indicative of stream condition and environmental health.

Research has tried to better quantify the effects of anthropogenic land use on salamander abundance,

density and most recently, occupancy. In the eastern United States, majority of this work has focused on the effects of forestry practices in relation to salamander abundances. In the central and southern Appalachians, data suggest forestry practices at the stand and landscape scale, at least in the short-term, lead to declines in terrestrial salamander abundance until sufficient forest maturation occurs (Ash, 1988; Petranka et al., 1993; Harper and Guynn, 1999; Ford et al., 2002). At smaller spatial scales, logging-related alterations to riparian quality (reduced canopy cover, leaf litter depth and decreased soil moisture) along with the alteration of physical stream conditions (change in stream substrate class, substrate embeddedness, riffle/run/pool composition) negatively influence stream salamander populations (Crawford and Semlitsch, 2008; Moseley et al., 2008; Peterman and Semlitsch, 2009). Stream salamanders may be significantly affected by fine spatial scales (microhabitat) more than landscape-level land use; however, microhabitat conditions are often directly related to cumulative historical and current watershed land use (Bury and Corn, 1989; Russell et al., 2004; Surasinghe and Baldwin, 2014). These studies illustrate the need for better understanding the dynamics among salamander occupancy, large-scale land use, and microhabitat, especially for Appalachian salamanders given the limited data available.

There has been minimal effort to examine the effects of coal mining on stream salamanders relative to forestry research. Most of the amphibian research on mined lands has focused on constructed settling ponds and use and recolonization by pond salamanders such as those in the family Ambystomidae (Fowler et al., 1985; Lacki et al., 1992; Jansen et al., 2004). Other work has centered on acid mine drainage because of its direct effects on stream biota, including stream salamanders (Freda, 1986; Middlekoop et al., 1998; Schorr et al. 2013). However, acid mine drainage is often a localized issue rather than pervasive throughout all the Appalachian coalfields (Herricks and Cairns, 1974; Minear and Tschantz, 1976).

Salamander community richness and abundance is generally lower in valley fill streams (aquatic systems buried by surface overburden during the mining process) than reference streams in central Appalachia (Hamilton, 2002; Wood and Williams, 2013a; Muncy et al., 2014). However, Hamilton (2002) observed no difference in 18 year old valley fill streams versus unmined reference streams. Nonetheless, complete examination linking microhabitat to landscape-level conditions and stream

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salamanders has rarely occurred. The authors' main objective was to investigate the relationship between salamander communities and habitat parameters at both a microhabitat and a landscape-level along a gradient of conditions in the coalfields of central Appalachia. These foundational correlations with an information-theoretic occupancy/detection modeling approach were examined using microhabitat data collected in the field along with a geographic information system (GIS) analysis of landscape-level land use.

MATERIALS AND METHODS

In southwest Virginia, five 12-digit Hydrologic Unit Codes (HUC-12) watersheds of similar area and located in the Cumberland Plateau and Mountains Range (Ecoregion 69d) of the central Appalachians were selected. Callahan Creek (area = 54.7 ha), Pigeon Creek (area = 58.9 ha), Roaring Fork (area = 66.0 ha) and Rock Fork (area = 91.0 ha) watersheds are located in Wise County, Virginia. Dumps Creek (area = 82.3 ha) is located in Dickenson and Russell Counties, Virginia. These watersheds have been substantially altered from past coal mining and residential areas as well as being actively mined for coal. The forested areas of the watersheds are characterized by a diverse mix of hardwood and conifers (Woodward and Hoffman, 1991). First- or second-order stream segments were randomly selected for salamander sampling sites; however, in some cases, best professional judgment was used to select sites that had landowner access and that considered safe to sample (Sweeten 2015). In total, 70 sites were sampled within the five HUC-12 watersheds. Because of the difficulty in accessing sites, the number of sites within each HUC-12 was not equal: 20 sites were sampled within Callahan Creek, 15 sites in Roaring Fork, 15 sites in Rocky Fork, 10 sites in Pigeon Creek, and 10 sites in Dumps Creek.

The authors attempted to sample each site three times between May 15 and August 15, 2013. However, because of access issues, three sites were only sampled twice in 2013. Two quadrats (25 m long by 5 m wide) were placed parallel to the stream at every sampling location with the edge of each quadrat placed along the center of the stream (Figure 1). At each sampling event, salamanders were captured from either the left or right quadrat (facing upstream) which was determined randomly. All rocks, logs and detritus within the quadrat were overturned in order to capture adult salamanders (all transformed salamanders, sexually mature and immature, were considered adults). Dip nets were used to capture larval salamanders within the stream. adult salamanders were identified to the species level, and larval salamanders to the genus-level. All salamanders were then released within a meter of their capture location.

Microhabitat parameters

The two 25 m by 5 m quadrats were combined for measuring riparian conditions as well as physical and chemical stream attributes. Past research was reviewed to identify microhabitat parameters thought to influence salamander occupancy and/or abundance (Bury and Corn, 1989; Miller et al., 2007; Moseley et al., 2008; Kroll et al., 2010). Temporally variable parameters such as water temperature, conductivity, soil temperature, air temperature and stream flow condition were sampled at each of the three sampling events. Parameters thought to likely remain constant over

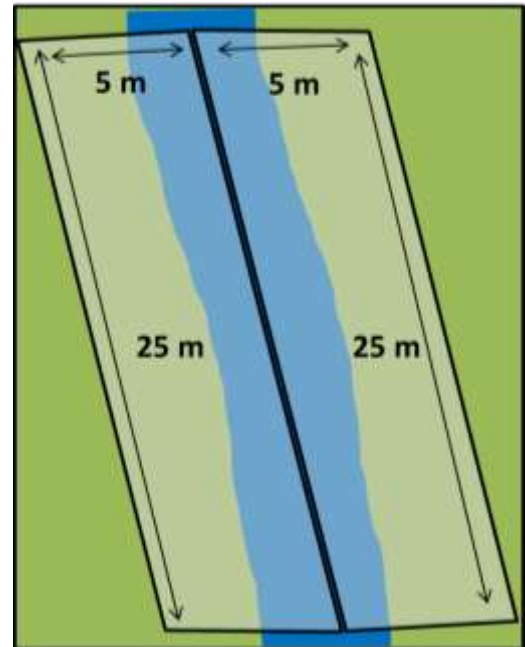


Figure 1. Salamander and riparian habitat quadrat placement used in southwest Virginia, 2013. Salamander sampling was done on either the left or right quadrat facing upstream (determined by a coin flip at each visit). Habitat measurements were taken from the full 25 x 10 m area.

the field season such as percent canopy cover, tree species richness and large woody debris (LWD) counts, were measured once during the summer. More details on collection methods for all microhabitat parameters are shown in Appendix 1.

Spatial analysis

Using multiple GIS layers such as the 2011 National Agriculture Imagery Program (NAIP), mining GIS data from the Virginia Department of Mines Mineral and Energy (DMME), the 2011 National Land Cover Dataset (NLCD), the USGS National Elevation Dataset (NED), the USGS National Hydrological Dataset (NHD), historical topographical maps and land use cover data from Maxwell et al. (2014), the authors measured landscape-level parameters at the subwatershed (the watershed from above the sampling location) scale and HUC-12 watershed scale. Parameters included watershed area, length of stream segment, kilometers of roads, area of valley fill, and number of structures and constructed ponds within the subwatershed. Land use was classified as percent of each watershed with recent mining (active mining permits and areas still barren), past mining and forested. All GIS analysis were performed using ArcMap² (v10.1, ESRI, Redlands, California). More details on the methods for the GIS analysis of landscape-level parameters are shown in Appendix 2.

Statistical analysis

The Program PRESENCE software (Any use of trade, firm or

Table 1. Detection covariates and data type of covariate used in occupancy analysis of stream salamanders in southwest Virginia, 2013.

Detection covariate	Covariate name	Covariate type	Additional description
Stream flow above base flow	Flow Above	Binomial	1 = Above 0 = Not above
Stream flow below base flow	FlowBelow	Binomial	1 = Below 0 = Not below
Soil temperature	SoilT	Continuous	°C
Air temperature	AirT	Continuous	°C
Water temperature	WaterT	Continuous	°C
Current weather rainy	Rain	Binomial	1 = Rainy 0 = Not rainy
Rain in Past 24 h	Rain24	Binomial	1= Rainy 0 = Not rainy

product names is for descriptive purposes only and does not imply endorsement by the U.S. Government) (v7.3, MacKenzie et al., 2006) was used to assess salamander occupancy and detection probabilities in an information-theoretic framework. Program PRESENCE fits multiple models and Akaike's Information Criterion (AIC) values and maximum likelihood were used to rank all detection (p) and occupancy (ψ , Ψ) parameters relative to environmental condition (Kroll et al., 2010). If detection is unaccounted for, the true presence of a species may be misclassified as absent if a species was present but not detected. Consequently, occupancy estimates can be skewed (MacKenzie, 2006).

Model selection

A two-step method was used to determine which detection covariates to include for each species in the occupancy analysis (Burnham and Anderson, 2002). *A priori* detection covariates was ran for each species against the null (intercept) model. All detection covariates that had an AIC smaller than the null were then run for all combinations to determine the best detection covariate for each species. This detection covariate was then used for the occupancy analysis. Utilizing this two-step method helps to reduce the total number of models in the final analysis. For this study, seven detection covariates were analyzed (Table 1). Prior to analysis, continuous detection covariates was normalized. 18 *a priori* models (containing 21 parameters) were developed for microhabitat occupancy analysis (Table 2). 15 landscape-level *a priori* models were developed using 14 GIS-derived covariates (Table 3). For both sets of models, covariates were grouped based on biologically relevant information or recommendations from available literature (Petranka et al., 1993; Ford et al., 2002; Willson and Dorcas, 2003; Moseley et al., 2008). All combinations of the parameters were not considered as this would have created unreliable outputs (Burnham and Anderson, 2002). Prior to analysis, all continuous occupancy covariates were normalized (Sweeten, 2015).

RESULTS

Species selection

Overall, nine species of aquatic salamanders were

detected during the 207 surveys (Table 4). Because of the large number of *Eurycea* spp. larval salamanders (442 individuals) and the small number of adult *Eurycea* spp. (39 individuals), larval and adult *Eurycea longicauda* (Long-tailed Salamander) and *Eurycea bislineata cirrigera* (Southern Two-lined Salamander) salamanders were combined to the genus-level for the *Eurycea* spp. group. All the larval and adult *Desmognathus* were also combined to the genus-level to make the *Desmognathus* spp. group. Based on estimated occupancy and detection probabilities, five salamander groups had sufficient data for occupancy analysis: *Desmognathus fuscus* (Northern Dusky Salamander), *Desmognathus monticola* (Seal Salamander), *Desmognathus ochrophaeus* (Mountain Dusky Salamander), *Desmognathus* spp. and *Eurycea* spp. (Table 5). However, the model fit (c -hat) for the *Desmognathus* spp. group was over-dispersed, and therefore, excluded from analysis. Species or genus groups that were detected too infrequently to be included in the occupancy analysis were *Desmognathus welteri* (Black Mountain Salamander), *Desmognathus* spp. larval, *Eurycea* spp. adults, *Eurycea bislineata cirrigera*, *Eurycea longicauda*, *Gyrinophilus porphyriticus* (Spring Salamander), *Pseudotriton montanus* (Mud Salamander) and *Pseudotriton ruber* (Red Salamander) (Table 4).

Occupancy results

In comparison of microhabitat and landscape-level models, the models with either strong empirical support (within Δ AIC of < 2) or moderate empirical support (Δ AIC of $2 - 4$) for *D. fuscus* were all microhabitat models (Table 6). The canopy model showed the most empirical support with an AIC weight of 0.2994 (Table 6). The percent canopy cover covariate was in five of the six top models for *D. fuscus* (Table 7). With the exception of the

Table 2. Final *a priori* occupancy models for microhabitat covariates used to examine the influence of fine-spatial scale conditions on stream salamanders, southwest Virginia, summer 2013.

S/N	Predictive model	Microhabitat parameters
1.	Null	Intercept
2.	Microhabitat Global	All microhabitat parameters
3.	Size	Stream width; Maximum stream depth
4.	Stream Location	Stream aspect; Gradient; Elevation
5.	Macrohabitat	Aspect; Gradient; Elevation; Canopy cover
6.	Shelter	Detritus cover; LWD Count; Stream bank erosion
7.	Stream composition	Percent pool
8.	Predation	Fish presence
9.	Chemistry	Conductivity; Water temperature
10.	Sediment	Turbidity; Stream embeddedness; Stream bank erosion
11.	Substrate	Mean substrate size; % Cobble and boulder; Stream embeddedness
12.	Canopy	Canopy cover
13.	Trees	Tree species richness; DBH; Canopy cover
14.	Herbaceous	Herbaceous cover; Saplings/shrubs
15.	Shading	Canopy cover; Herbaceous cover; Sapling/shrubs; DBH
16.	Detritus	Detritus cover
17.	Future detritus	Tree snag count; Canopy cover
18.	Cover	LWD count; Detritus cover; % Cobble and boulder; Stream bank erosion

Table 3. *A priori* occupancy models with large-scale land use covariates used to examine the influence of landscape-level conditions on stream salamanders, southwest Virginia, summer 2013.

S/N	Predictive models	Large-scale parameters
1.	Null	Intercept
2.	Global	All landscape-level parameters
3.	Forested	Subwatershed forested
4.	Residential	Structures
5.	Non-Mining	Structures; Roads
6.	Past Mining	Subwatershed past mining
7.	Recent Mining	Subwatershed recent mining
8.	Surface Mining	Subwatershed recent mining; Valley fill; Ponds in segment; Ponds downstream
9.	Valley Fill	Valley fill
10.	Pond	Ponds in segment; Ponds downstream
11.	Mining Streams	Valley fill; Ponds in segment
12.	HUC-12 Forested	HUC-12 watershed forested
13.	HUC-12 Past Mining	HUC-12 watershed past mining
14.	HUC-12 Recent Mining	HUC-12 watershed recent mining
15.	Physical	Subwatershed area; Stream order; Receiving stream order

Non-Mining Model (a landscape-level model) with a Δ AIC of 2.78, all models with empirical support (within Δ AIC of 4) for *D. monticola* were models with microhabitat or small-scale components (Table 6). The Shelter Model was the top ranked model with an AIC weight of 0.2895 (Table 6). The covariates percent detritus cover and the

number of large woody debris pieces (LWD) were present in two of the top three models, and were positively associated with *D. monticola* occupancy (Table 7). *D. ochrophaeus* results had empirical support for microhabitat models (Table 6). The best-ranked model was the Macrohabitat Model with an AIC weight of 0.4202

Table 4. Number of sampling sites and sampling events in which salamanders were found in summer 2013, southwest Virginia.

Scientific name	Sampling sites (n = 70)	Sampling events (n = 207)
<i>Desmognathus fuscus</i>	27	45
<i>Desmognathus monticola</i>	40	81
<i>Desmognathus ochrophaeus</i>	37	75
<i>Desmognathus welteri</i>	18	21
<i>Desmognathus</i> Larval	6	6
<i>Eurycea b. cirrigera</i>	15	19
<i>Eurycea longicauda</i>	7	8
<i>Eurycea</i> Larval	42	85
<i>Gyrinophilus porphyriticus</i>	10	11
<i>Pseudotriton montanus</i>	3	3
<i>Pseudotriton ruber</i>	6	6

Table 5. Observed occupancy (naïve occupancy), model-averaged estimations of occupancy (Ψ) and detection (p) along with standard errors for the four species/genus groups (*Desmognathus fuscus*, *Desmognathus monticola*, *Desmognathus ochrophaeus* and *Eurycea* spp.) of stream salamanders used for full analysis of microhabitat and landscape-level models, southwest Virginia, 2013.

Species group	Observed Ψ	Estimate of Ψ	Ψ SE	Estimate of p	p SE
<i>D. fuscus</i>	0.3857	0.4507	0.0743	0.4824	0.0701
<i>D. monticola</i>	0.5714	0.6002	0.0633	0.6494	0.0496
<i>D. ochrophaeus</i>	0.5286	0.5486	0.0627	0.6685	0.0508
<i>Eurycea</i> spp.	0.7000	0.7262	0.0577	0.6787	0.0432

Table 6. Top models for occupancy estimates including the number of parameters in each model (K), Akaike's Information Criterion (AIC) rankings, Δ AIC and AIC weight (ω_i) for the four groups of stream salamanders, southwest Virginia, summer 2013.

Occupancy models	K	AIC	Δ AIC ^a	ω_i
<i>Desmognathus fuscus</i>				
Ψ (Canopy), p (Rain24) ^b	4	182.44	0.00	0.2994
Ψ (Trees), p (Rain24)	6	183.49	1.05	0.1771
Ψ (FutureDetritus), p (Rain24)	5	183.53	1.09	0.1736
Ψ (Shading), p (Rain24)	7	184.68	2.24	0.0977
Ψ (Detritus), p (Rain24)	4	184.94	2.50	0.0858
Ψ (Macrohabitat), p (Rain24)	7	185.58	3.14	0.0623
Ψ (Shelter), p (Rain24)	6	186.50	4.06	0.0393
<i>Desmognathus monticola</i>				
Ψ (Shelter), p (WaterT) ^c	6	223.77	0.00	0.2895
Ψ (Macrohabitat), p (WaterT)	7	224.07	0.30	0.2492
Ψ (Cover), p (WaterT)	7	225.06	1.29	0.1519
Ψ (Canopy), p (Water)	4	226.44	2.67	0.0762
Ψ (NonMining), p (Rain24) ^f	5	226.55	2.78	0.0721

Table 6. Contd.

Ψ (Trees), p(WaterT)	6	226.60	2.83	0.0703
Ψ (Shading), p(WaterT)	7	227.75	3.98	0.0396
Ψ (FutureDetritus), p(WaterT)	5	228.44	4.67	0.0280
<i>Desmognathus ochrophaeus</i>				
Ψ (Macrohabitat), p(SoilT*Rain) ^{d,e}	8	209.92	0.00	0.4202
Ψ (Shading), p(SoilT*Rain)	8	211.93	2.01	0.1538
Ψ (Trees), p(SoilT*Rain)	7	212.01	2.09	0.1478
Ψ (Canopy), p(SoilT*Rain)	5	212.83	2.91	0.0981
Ψ (Chemistry), p(SoilT*Rain)	6	213.45	3.53	0.0719
Ψ (FutureDetritus), p(SoilT*Rain)	6	214.76	4.84	0.0374
<i>Eurycea</i> spp.				
Ψ (HUC12RecentMining), p(Rain) ^f	4	249.84	0.00	0.5599
Ψ (HUC12Forested), p(Rain) ^f	4	251.13	1.29	0.2938
Ψ (NonMining), p(Rain) ^f	5	255.24	5.40	0.0376

^a Models with a Δ AIC < 2 are considered to have a substantial level of empirical support. Models with a Δ AIC of 2 – 4 are considered to have a moderate level of empirical support. ^b Rain 24 is a binomial for weather in past 24 h rainy. ^cWaterT is water temperature. ^dSoilT is soil temperature. ^eRain is a binomial for current weather rainy. ^fLandscape-level model.

(Table 6), and percent canopy cover covariate had a positive relationship with *D. ochrophaeus* occupancy (Table 7). Landscape-level models had empirical support for *Eurycea* spp. (Table 6). The HUC-12 Recent Mining Model is the top model with an AIC weight of 0.5599 (Table 6). Within this model, the percent of recent mining within the HUC-12 watershed showed a negative correlation with *Eurycea* spp. occupancy probabilities (Table 8).

DISCUSSION

It was found that *Desmognathus* spp. had similar results with most top models being microhabitat models. Within the microhabitat models, most top models (within a Δ AIC of 4) contained canopy cover as a covariate. Five occupancy models best explained presence of the three *Desmognathus* species. These models were the Canopy Model, the Trees Model, the Shading Model, the Shelter Model and the Macrohabitat Model. Multiple parameters comprised these models including canopy cover, tree diameter at breast height (DBH), tree species richness, herbaceous cover, sapling/shrub density, detritus cover, LWD and stream bank erosion. Because data were normalized prior to analysis, it was possible to compare the betas, or effect size, of these different parameters and examine correlation trends among models and species. Canopy cover was a parameter in four of the five models.

All the three *Desmognathus* species were positively

correlated with increased canopy cover. Additionally, beta values for canopy cover in the top models were large, indicating a strong effect size. Past research has also shown strong associations between canopy cover and salamander abundance (Davic and Welsh, 2004; Crawford and Semlitsch, 2008; Ward et al., 2008). *Desmognathus* salamanders are lungless and therefore are required to constantly have moist skin in order for oxygen exchange across the skin membrane (Petranka, 1998). Lungless salamanders have a high risk of desiccation particularly when foraging away from the stream. Canopy cover not only provides cover from solar exposure lowering soil, stream, and air temperatures, it also increases other microhabitat parameters associated with maintaining a cool, moist environment such as detritus cover. Crawford and Semlitsch (2008) found a positive correlation in riparian areas between detritus depth and *D. monticola* and *E. b. cirrigera*. In upland plots in the southern Appalachians of North Carolina, Harper and Guynn (1999) observed more salamanders including *D. ochrophaeus* and *D. aeneus* in moist microhabitats with increased detritus depths.

A closed canopy cover is also often an indication of mature, less-disturbed or more recovered forest conditions. Although, forest stand age was not a covariate in this study, *Desmognathus* spp. were positively correlated with characteristics often found in more mature forests such as high canopy cover, native tree species dominance, high detritus cover and more LWD. In nearby Ecoregions 67 and 69 of West Virginia, Moseley et al. (2008) found a positive relationship between *Desmognathus*

Table 7. Beta estimates and standard errors for each covariate in the top occupancy models (Δ AIC < 2) for the three *Desmognathus* species, southwest Virginia, summer 2013.

Occupancy models and parameters	Beta	Standard error
<i>Desmognathus fuscus</i>		
Ψ(Canopy), p(Rain24)		
Ψ .CanopyCover	1.8792	0.7157
p.Rain24	-1.0272	0.5211
Ψ(Trees), p(Rain24)		
Ψ .TreeSppRichness	0.6642	0.4594
Ψ .TreeDiameter	0.1906	0.3930
Ψ .CanopyCover	1.1249	0.7442
p.Rain24	-1.0185	0.5148
Ψ(FutureDetritus), p(Rain24)		
Ψ .TreeSnags	-0.2925	0.3213
Ψ .CanopyCover	2.0285	0.7642
p.Rain24	-1.0264	0.5178
<i>Desmognathus monticola</i>		
Ψ(Shelter), p(WaterT)		
Ψ .DetritusCover	1.0607	0.4178
Ψ .LWD	0.6710	0.6664
Ψ .StreamBankErosion	1.0507	0.8032
p.WaterT	0.3000	0.3892
Ψ(Macrohabitat), p(WaterT)		
Ψ .Aspect	0.3168	0.3969
Ψ .Gradient	1.6145	0.7554
Ψ .Elevation	-0.0910	0.4254
Ψ .CanopyCover	1.9307	0.5948
p.WaterT	0.5503	0.3114
Ψ(Cover), p(WaterT)		
Ψ .Cobble/Boulder	-0.2582	0.4073
Ψ .LWD	2.2038	1.0469
Ψ .DetritusCover	1.0890	0.4445
Ψ .StreamEmbeddedness	1.0401	0.4963
p.WaterT	0.4790	0.2785
<i>Desmognathus ochrophaeus</i>		
Ψ(Macrohabitat), p(SoilT*Rain)		
Ψ .Aspect	0.6174	0.3328
Ψ .Gradient	-0.0700	0.3077
Ψ .Elevation	0.7126	0.3504
Ψ .CanopyCover	1.2020	0.3729
p.SoilT	-0.5662	0.2537
p.Rain	-1.9292	0.7413

spp. abundance in first- and second-order streams and time since forest harvest in the immediate watershed. Ford et al. (2002) observed that *Desmognathus* spp. abundance generally was most correlated to tree basal area within forest stands in southern Appalachia. Additionally, in a review of North American literature on

amphibian ecology and forest management, deMaynadier and Hunter (1995) suggested increased salamander abundance in older forests functionally are an indirect measure of microhabitat conditions such as LWD, detritus cover and canopy cover.

Results from the present study show that *Desmognathus*

Table 8. Beta estimates and standard errors of the covariates in the top occupancy models (Δ AIC < 2) for *Eurycea* spp., southwest Virginia, summer 2013.

Occupancy models and parameters for <i>Eurycea</i> spp.	Beta	Standard error
Ψ (HUC12RecentMining), p(Rain)		
Ψ .HUC12RecentMining	-1.4876	0.6450
p.Rain	1.0503	0.6620
Ψ (HUC12Forested), p(Rain)		
Ψ .HUC12Forested	2.6829	1.9141
p.Rain	1.0961	0.6578

were negatively correlated with parameters associated with open areas/grasslands such as increased herbaceous cover and areas dominated by invasive species. In Maryland, Walz (2002) found decreased abundances of *D. fuscus* and *D. ochrophaeus* in agricultural fields and pastures. Wood and Williams (2013b) found lower abundances of *Desmognathus* in reclaimed grassland and shrublands in the West Virginia coalfields (Ecoregion 69) where there was less detritus, lower stem densities, less LWD, less canopy cover, and an increase in invasive herbaceous species such as *Lespedeza* spp. as compared to forested or partially forested sites. Invasive herbaceous species could be indicative of recent disturbance, or it may be that established invasive plant species simply do not produce the necessary microhabitat (leaf litter, cover, and LWD) to provide the cool, moist habitat needed for salamanders (Lemke et al. 2012).

With the exception of the Non-Mining Model for *D. monticola*, large-scale land use models were not supported for *Desmognathus* spp. occupancy. This indicates that local riparian areas are more important for *Desmognathus* spp. than watershed or subwatershed land uses. In the Ecoregion 69 portion of Kentucky, Maigret et al. (2014) found that *Desmognathus* spp. abundances were lower in areas without a timber harvest stream buffer zone than in undisturbed areas or areas with a 7.6 m stream buffer zone. Pearl et al. (2005) found that in the Pacific Northwest wetland and riparian conditions are most influenced by native salamander species occupancy and land use within a 1000 m radius had little effect on salamander occupancy.

The Non-Mining Model for *D. monticola* was the only large-scale land use with empirical support for the *Desmognathus* spp. This model consists of the number of houses and total km of roads in the subwatershed. Beta results show that the kilometers of road had a much larger effect size than the number of houses. Similarly, Ward et al. (2008) also found a negative correlation between *Desmognathus* spp. abundance and roads in central Appalachia. Roads often tend to cause forest fragmentation, which decreases canopy cover and

detritus (Ward et al., 2008). Additionally, deMaynadier and Hunter (2000) found that forested sites without roads generally had twice the number of salamanders than roadside sites. They suggested that movement of both terrestrial and aquatic salamanders may be limited by roads. Impassible culverts can impede upstream salamander movement, and may prevent migration altering salamander community structure upstream of road-stream crossings (Ward et al., 2008; Anderson et al., 2014).

Roads, particularly paved roads, may also alter water chemistry parameters such as conductivity. Runoff from roads containing de-icing salt has been shown to travel over 170 m from the road into wetlands in the Adirondack Mountains of New York (Karraker et al., 2008). In vernal pools within 50 m of roads, Karraker et al. (2008) reported declines in *Ambystoma maculatum* (Spotted Salamander) abundance thought to be caused by increased salinity from road salt. Nonetheless, Jones et al. (2015) found no differences in the toxicity of road salt versus NaCl to *Rana clamitans* (Green Frog) or *E. bislineata*. They also found no mortality for *E. bislineata* from conductivity until concentrations reached ~10,000 us/cm (Jones et al., 2015). Additionally, Izzo (2013) found that road salt had no lethal effect on *D. fuscus* or *E. bislineata* until chloride concentrations exceeded 5,000 mg/l (~9,000 us/cm). The conductivity range for this study was 13-1660 us/cm. This study did not find conductivity (in the Chemistry Model) to have a strong relationship with salamander occupancy. The Chemistry Model only had weak empirical support for one species, *D. ochrophaeus*, with a Δ AIC of 3.53.

Eurycea spp. occupancy results showed that occupancy was negatively associated with the percent mining in the whole HUC-12 watershed, whereas the percent forested in the HUC-12 watershed was positively correlated with *Eurycea* spp. occupancy. Additionally, microhabitat models for *Eurycea* spp. did not have any empirical support. These results also found that other than the Recent Mining HUC-12 Model, none of the other potential mining-related parameters had empirical support (e.g. the Chemistry Model, the Valley Fill Model, and the Surface

Mining Model) for *Eurycea* spp. In the Piedmont of North Carolina, Willson and Dorcas (2003) found that upland watershed disturbances (> 20% of watershed), such as agriculture and residential areas, caused declines in *Eurycea* (larval and adult) abundances, but that riparian buffer size and quality was not correlated to relative abundance. Similarly, Miller et al. (2007) found in the Piedmont of North Carolina that larval *E. cirrigera* abundances were negatively associated impervious surfaces in the watershed from residential and urban areas, however, abundances were not influenced by 50 m forested riparian buffers.

This difference in response of *Eurycea* spp. and *Desmognathus* spp. to microhabitat and landscape-level land use may be a function of movement and habitat selection. Adult *Eurycea* spp. are more terrestrial than the *Desmognathus* species, and *E. b. cirrigera* have been shown to migrate over 100 m away from the stream, whereas *D. fuscus* are largely sedentary having small home ranges of 1.4 to 3 m² (Barthalmus and Bellis, 1972; Ashton, 1975; MacCulloch and Bider, 1975). Larval *E. bislineata* have been shown to move in stream. Small first-year larval tend to drift downstream, whereas larger second-year larval will move both upstream and downstream equally (Bruce, 1986; Petranka, 1998). The ability of *E. bislineata* to move long-distance may also explain why recent mining in the HUC-12 watershed had more of an influence on *Eurycea* spp. occupancy than recent mining in the subwatershed.

Predation and competition may also contribute to differences in results between *Eurycea* spp. and *Desmognathus* spp. Predation may have both direct mortality and indirect behavioral effects on stream salamanders, particularly on *Eurycea* spp., the smallest stream salamander regionally with the most potential aquatic predators. For example, Barr and Babbitt (2002) found that densities of *E. b. bislineata* were dramatically lower (9.4/m² as compared to 54/m²) when the predatory *Salvelinus fontinalis* (Brook Trout) was present. However, *Eurycea* spp. may be more susceptible to predation from other stream salamanders than from fish predation. A stomach content analysis showed that up to 50% of *G. porphyriticus*, and 18% of *D. quadramaculatus* (Black-bellied Salamander) stomachs contained larval *E. b. cirrigera* (Bruce 1972). Predation from other stream salamanders may not only decrease survival rates, but may also influence microhabitat selection and behavior. Without predators, *E. b. wilderae* (Blue Ridge Two-lined Salamander) were evenly distributed in pools and riffles, however, when *D. quadramaculatus*, a large predator species that prefers riffles, were present, *E. b. wilderae* were more abundant in pools than in riffles (Beachy, 1993). Keen (1982) found when *D. monticola*, a predator of *D. fuscus*, was present, activity levels of *D. fuscus* dropped significantly.

Others have also hypothesized that decreases in some salamander species abundances when predatory fish and/or salamander species are present may not come from predation but rather from competition (Hairston, 1980; Davic, 1983; Barr and Babbitt, 2002; Bruce, 2011). Small-bodied, generalist salamanders such as *Eurycea* spp. are opportunistic generalists with diets largely consisting of pollutant-tolerant benthic macro-invertebrates such as *Chironomids* (Burton, 1976; Petranka, 1984; Muenz et al., 2008; Barrett et al., 2012). *Eurycea* spp. is perhaps better able to tolerate poor water quality and riparian habitat conditions where prey items (*Chironomids*) are often readily available. These disturbed conditions not only provide a plentiful food source for *Eurycea* spp., but may also provide a competitive release from larger salamander such as *Desmognathus* spp. and *G. porphyriticus* (Barrett et al., 2012; Surasinghe and Baldwin, 2015). However, the presence of *Eurycea* spp. in disturbed areas may also be influenced by predatory release (Ransom and Jaeger, 2006). More research that limits natural variability is needed to examine interspecific competition and predation among salamander species.

Overall, this research found effects of mining on stream salamanders to be equivocal; however, this is not to imply that mining does not affect stream salamanders.

Although, the top model for *Eurycea* spp. was the HUC-12 Recent Mining Model, none of the species in this study had strong empirical support for many of the other mining-related parameters such as the percent active or past mining in the subwatershed, area of valley fill, conductivity, and the number of settling ponds. Past studies have found negative correlations between stream salamanders and coal mining (Hamilton, 2002; Wood and Williams, 2013a, b; Muncy et al. 2014). Still, these studies all examined the effects of mining versus reference conditions, and the resulting decreases in salamander assemblages may be a reflection of general riparian disturbance and not factors particularly unique to coal mining. The research helped differentiate effects of coal mining and general riparian disturbance on stream salamanders by using a gradient of land uses and disturbances. This gradient of sites showed poor riparian quality lead to declines in *Desmognathus* spp. abundance regardless of the type of large-scale land use associated with that riparian disturbance.

Although, stream salamander recovery from coal mining disturbance has not specifically been examined in this study, these results suggest that *Desmognathus* spp. occupancy may improve as riparian habitat quality increases post-mining. Proper mine land reclamation can decrease the amount of time needed for recovery. Currently, most mine land reclamation in central Appalachia has two terminal land uses: reforestation or grass-shrubland. Reforestation techniques such as those

recommended by Appalachian Regional Reforestation Initiative (ARRI) suggest planting high-value hardwood trees on reclaimed mine land in order to produce commercially valuable crop trees along with quick-growing early successional trees for soil stability and initial wildlife value. Currently, the ARRI protocol does not have different management plans for riparian areas and upland areas. Additionally, current regulatory reclamation standards for grassland-shrubland land uses do not require reforestation of riparian areas. Regardless of target land use, in order to promote stream salamander recovery following mining, specific riparian reclamation guidelines are warranted. For example, quick-growing early successional tree species such as *Betula lenta* (Black Birch), *Liriodendron tulipifera* (Yellow Poplar), *Salix nigra* (Black Willow), *Platanus occidentalis* (American Sycamore), *Populus deltoids* (Eastern Cottonwood) and *Pinus strobus* (White Pine) would quickly establish a canopy cover, stabilize the soil, and impede invasive plant species (Davis et al., 2012) for more extensive list of suitable tree species adapted for moist/wet sites. Because, timber harvest within a riparian zone has been shown to cause changes in benthic macroinvertebrate, stream salamander, and fish densities and community structure (Jones et al., 1999; Rios and Bailey, 2006; Maigret et al., 2014), emphasis on riparian tree species prioritization could be based on those with soil retention/development and/or wildlife value instead of crop tree value. By developing a differential reforestation management plans for upland and riparian areas, a closer approximation of a native Appalachian forest may be achieved more quickly, and in turn provide high-quality refuges for stream salamanders and other biotic components of these aquatic ecosystems.

Conflict of Interests

The authors have not declare any conflict of interestS.

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Appendix 1. Parameters measured in the field, southwest Virginia, summer 2013.

Field parameters	Units	Sampling events	Additional description
1. Fish Presence	Binomial	3	1 = Observed; 0 = Not observed
2. Conductivity	µs/cm	3	Hach water meter
3. Turbidity	cm	3	Depth of secchi tube visibility
4. Water Temperature	°C	3	Measured in shade
4. Stream Width	m	1	Mean of 3 wetted stream width measurements
5. Stream Depth	cm	1	Maximum depth at base flow
6. Aspect	Degree	1	Compass reading of stream aspect
7. Gradient	Percent	1	Clinometer reading from bottom of quadrat to top of quadrat
8. Canopy Cover ^a	Percent	1	Mean of 3 convex densitometer measurements
9. Detritus Cover ^a	Percent	1	Mean of 6 estimations
10. Large Woody Debris ^a	Count	1	Count of LWD
11. Stream Bank Erosion ^a	Percent	1	Mean of left and right banks erosion estimations
12. Pool Composition	Percent	1	Estimation of percent of quadrat comprised of pool habitat
13. Substrate Size ^a	cm	1	Mean of 30 measurements
14. Stream Embeddedness ^a	Score 1-5	1	Mean of 30 estimations
15. Tree Species Richness ^a	Score 1-5	1	Ratio of native tree spp. and invasive plant spp.
16. Tree Diameter (DBH) ^a	Count	1	Number of trees with DBH > 10 cm
17. Herbaceous Cover ^a	Percent	1	Mean of 6 estimates
18. Sapling/Shrub ^a	Count	1	# Stems with DBH < 10 cm
19. Tree Snags ^a	Count	1	# Standing, dead trees
20. Cobble/Boulder	Percent	1	Estimate of stream substrate cobble and boulder composition

^a See Noble et al. (2010) for sampling protocol details.

Appendix 2. Parameters derived from GIS analysis, southwest Virginia, summer 2013 (see Sweeten 2015 for additional details).

Landscape covariate	Unit	GIS data used
Forested ^a	Percent	Maxwell et al., 2014
Past mining ^a	Percent	Maxwell et al., 2014; NAIP, 2011; DMME, 2013; NLCD, 2011
Recent mining ^a	Percent	Maxwell et al., 2014; NAIP, 2011; DMME, 2013; NLCD, 2011
Structures ^a	Count	NAIP, 2011
Roads ^a	Kilometers	NAIP, 2011
Valley fill ^a	Hectares	Maxwell et al., 2014; DMME, 2013
Ponds in segment ^a	Count	NAIP, 2011
Ponds downstream	Count	NAIP, 2011
HUC-12 forested	Percent	Maxwell et al., 2014
HUC-12 past mining	Percent	Maxwell et al., 2014; NAIP, 2011; DMME, 2013; NLCD, 2011
HUC-12 recent mining	Percent	Maxwell et al., 2014; NAIP, 2011; DMME, 2013; NLCD, 2011
Subwatershed area	Hectares	NED
Stream order		NHD
Receiving stream order		NHD
Elevation ^b	Meters	NED

^a Calculated within subwatershed from the salamander sampling location upstream. ^b Elevation was used for microhabitat models. All other covariates in this table were used in the landscape-level analysis.