



Mountaintop removal mining reduces stream salamander occupancy and richness in southeastern Kentucky (USA)



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ABSTRACT

Mountaintop removal mining with valley fills (MTR/VF) is a ubiquitous form of land conversion in central Appalachia, USA and threatens the integrity of stream ecosystems. We investigated the effects of MTR/VF on stream salamander occupancy and overall community composition in southeastern Kentucky by conducting area constrained active searches for salamanders within first-order streams located in mature forest (i.e., control streams) and those impacted by MTR/VF. We found high mean species occupancy across 5 species at control streams, ranging from 0.73 (95% CI 0.41 to 0.96) to 0.90 (95% CI 0.77 to 0.98). Occupancy was lower at MTR/VF streams, with mean estimated occupancy probability ranging from 0.23 (95% CI 0.04 to 0.51) to 0.62 (95% CI 0.36 to 0.86). Additionally, the mean species richness for MTR/VF streams was 2.27 (± 1.27 SD) whereas richness was 4.67 (± 0.65 SD) for control streams. Numerous mechanisms may be responsible for decreased occupancy and species richness at MTR/VF streams, although water chemistry may be particularly important. Indeed, mean specific conductance was 30 times greater, sulfate (SO₄) levels were 70 times greater, and concentrations of dissolved ions (Ca, Mg, K, Na) were greater in MTR/VF streams than in control streams. Our results indicate that MTR/VF operations lead to significant decreases in salamander occupancy and species richness.

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1. Introduction

Understanding the environmental impacts associated with mining has become an issue of global importance (Cooke and Johnson, 2002; Bridge, 2004; Litz et al., 2013). In central Appalachia (USA), mountaintop removal mining, a form of surface mining, has become the primary method for coal extraction. The coal seams are accessed by first removing forests, then clearing and stripping topsoil, and finally, using explosives, overlain rocks are removed to allow for excavation of coal (Palmer et al., 2010). The overburden material that is removed (i.e., mine “spoil”) is pushed into an adjacent valley, burying portions of ephemeral, intermittent, and perennial streams located next to mining operations and creating a valley fill (Bernhardt and Palmer, 2011). When exposed to atmo-

spheric conditions and surface runoff, the unweathered overburden material often leaches heavy metals along with high levels of salts into the partially buried streams (Griffith et al., 2012). Thus, water that emerges from the base of valley fills can exhibit altered pH, greater specific conductance, and elevated levels of total dissolved solids (i.e., sulfates (SO₄), calcium (Ca), magnesium (Mg)) compared to unaltered streams (Fritz et al., 2010; Palmer et al., 2010; Barton, 2011; Lindberg et al., 2011). Additionally, because of reduced vegetative cover and highly compacted soils on mined lands, streams impacted by mountaintop removal mining with valley fills (MTR/VF) typically have altered hydrology (i.e., decreased infiltration, increased peak flows) compared to streams within forested catchments (Negley and Eshleman, 2006). More than 1.1 million ha of forest land has been altered by surface mining in central Appalachia, USA (Bernhardt and Palmer, 2011). In the Commonwealth of Kentucky, approximately 2000 km of streams have been impacted by valley fills (Barton, 2011), and over 20% of streams in southern West Virginia are affected by runoff from surface coal mines (Bernhardt et al., 2012).

Streams influenced by MTR/VF are often characterized by diminished biological communities in comparison to reference streams. For example, macroinvertebrate richness in MTR/VF

Abbreviations: MTR/VF, mountaintop removal mining with valley fills; Ψ , occupancy; Θ , detection probability; u_i , species-specific mean probability of occurrence; v_i , species-specific mean probability of detection; U, uniform distribution.

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streams is significantly reduced compared to reference locations (Pond, 2010, 2012), and freshwater mussel diversity decreases as extent of surface mines increase within catchments of central Appalachian rivers (Warren and Haag, 2005). Additionally, fish species richness is reduced by 50% at sites downstream from MTR/VF (Ferreri et al., 2004). Amphibians, specifically salamanders, are important components of low-order stream ecosystems (Davic and Welsh, 2004); up to 9 species occur within central Appalachian streams (Petranka, 1998). Salamanders represent the dominant predators in low-order streams, and are responsible for driving many ecosystem-level processes (i.e., nutrient cycling; Davic and Welsh, 2004; Keitzer and Goforth, 2013). Although Wood and Williams (2013a) documented reduced abundances of stream salamanders in MTR/VF streams, investigations on the responses of stream salamander species' occupancy and communities to MTR/VF are lacking.

To evaluate the effects of MTR/VF on stream salamanders, we compared species' occupancy and community composition within streams located in mature, second-growth forest (i.e., control streams) to MTR/VF streams located on reclaimed mountaintop removal mined land. Specifically, we employed a multi-species hierarchical model to estimate species-specific and community-level responses of salamanders to MTR/VF while accounting for species-specific variation in detectability (Zipkin et al., 2009; Hunt et al., 2013). Additionally, we evaluated water chemistry attributes and other habitat characteristics of MTR/VF and control streams to determine mechanisms potentially responsible for species' occupancy and community composition. We hypothesized that MTR/VF would have a negative effect on species' occupancy probabilities and richness, and that MTR/VF streams would differ significantly in water chemistry and habitat characteristics from control locations.

2. Methods

2.1. Study sites

We investigated salamander occupancy and community composition at 23 first-order streams located in the interior rugged section of the Cumberland Plateau in Breathitt and Knott Counties, Kentucky USA. This region has seen extensive changes in land-use over the last 30 years; more than 194,000 ha of eastern Kentucky has been affected by surface mining (C. Barton, personal communication). We sampled salamanders at 11 MTR/VF first-order streams located on the reclaimed Laurel Fork surface mine (4144091.438 N 307635.435 E Zone 17) and 12 control first-order streams in approximately 80-yr-old, second-growth forest on the University of Kentucky's Robinson Forest, which shares a northeast border with the Laurel Fork surface mine. Robinson Forest is a 5983 ha teaching, research and extension experimental forest composed of eight discontinuous properties. Our control streams were located with the main block of Robinson Forest comprising approximately 4200 ha. Land-cover within catchments of control streams consisted of typical, mixed mesophytic forests of the region; dominant tree species included white oak (*Quercus alba*), tulip tree (*Liriodendron tulipifera*), Eastern hemlock (*Tsuga canadensis*), and chestnut oak (*Quercus prinus*) (see Phillippi and Boebinger, 1986).

During the mid-1990s, approximately 607 ha of the 890 ha Laurel Fork watershed, was mined for coal. The catchments of the MTR/VF streams sampled in our study were mined in the late 1990s and reclamation occurred in the early 2000s. Bond release, indicating that reclamation was satisfied, was issued in November of 2007. All of the streams used in this study were partially buried by overburden (i.e., valley-filled); all VFs had perimeter drains, which collect seepage and runoff from around the VF and direct

the runoff into the original stream channel. Dominant vegetation cover of the MTR/VF catchments included the nitrogen-fixing herb *Sericea lespedeza* (*Lespedeza cuneata*) and grasses (tall fescue; *Schedonorus arundinaceus*), with autumn olive (*Elaeagnus umbellata*), Virginia pine (*Pinus virginiana*), white oak (*Q. alba*) and black locust (*Robinia pseudoacacia*) scattered throughout the landscape. Despite low forest cover within catchments, all MTR/VF stream riparian zones and adjacent terrestrial habitat was primarily forested. See Fritz et al. (2010) for additional information on the Laurel Fork study site.

2.2. Data collection methods

Area-constrained active searches were used to sample salamanders at each stream, in a single, 10-m sampling transect. Transects were chosen on the basis of similarity of width, depth and current velocity. Additionally, all transects included a pool, run and riffle section. Streams impacted by MTR/VF were generally sampled at the base of the VF. Although previous studies on stream salamanders have utilized longer transects (i.e., 100 m (Lowe et al., 2004)), the 10-m length of our sampling transect was chosen because of logistical reasons (i.e., dense salamander populations; large number of cover objects) and to provide data comparable to previous studies of stream salamander occupancy in the eastern US (i.e., Grant et al., 2009; Price et al., 2011).

We used a combination of systematic dipnetting and bank searches to capture salamanders (see Price et al., 2011). Dipnetting consisted of one person, moving from downstream to upstream, actively searching for salamanders around and under submerged rocks, logs, and other cover within the 10-m sampling transect. One person also conducted bank searches, which included searching under rocks, logs, leaf litter and other material within 1 m of the wetted width of the stream. In general, dipnetting sessions took approximately 30 min and bank searches took 15 min to finish. All salamanders captured were held in containers until the search was complete. After the active search, we recorded each species and the associated life stage (adult or larva) prior to release. Each 10-m transect was sampled four times (i.e., usually monthly) from March through June 2013. All searches were conducted during day light hours in base flow conditions.

We recorded several variables before each active search. Prior to sampling, we measured the wetted width and depth at the start, middle, and end of each 10 m sampling transect and counted the number of cover objects within the wetted width of our sampling transects. Specifically, we considered rocks >50 mm in diameter as well as logs and other debris cover objects of importance to salamanders. Also, we recorded air temperature (°C), water temperature (°C), wind speed, degree of cloudiness, and the date of last precipitation. Additionally, a 50 mL water sample was collected prior to each sampling event and placed on ice. The samples were analyzed for concentrations of Ca, Mg, SO_4^{2-} , potassium (K), sodium (Na), total organic carbon (TOC), pH and specific conductance; sampling, preservation, and analytic protocols were performed in accordance with standard methods (Greenberg et al., 1992).

Finally, we used a geographic information system (ArcGIS 10.1 ESRI) and Watershed tool in ArcToolBox to calculate the catchment area and percent of catchment in forest cover of each of study stream. To calculate catchment area, we used post-mining, high resolution (0.6 m), digital elevation model (DEM) data as our base layer for catchment delineation. Forest cover was obtained via 2013 United States Geological Survey 7.5-min image map for Noble, KY quadrangle; we considered both mature and younger forest classes as forest cover in our analysis of each stream catchment.

2.3. Data analysis

We used Bayesian t -tests with unequal variances (Kéry, 2010) to compare several environmental attributes between control and MTR/VF streams. Attributes included: percent of the stream catchment in forest cover, average stream wetted width and depth in our sampling transects, number of cover objects within our sampling transects, water temperature, specific conductance, TOC, pH, SO_4 , Ca, Mg, K, and Na. All water quality data used in the analysis were obtained during May 1–15, 2013 salamander sampling events. We used uninformative priors for each model, which varied depending on the covariate being analyzed (i.e., percent forest cover mean = Uniform distribution $U(0,1)$, standard deviation (SD) = $U(0,10)$; average stream width mean = $U(0,250)$, SD $U(0,300)$; average stream depth mean = $U(0,25)$, SD = $U(0,30)$; cover objects mean = $U(0,80)$, SD = $U(0,100)$; water temperature mean = $U(0,25)$, SD $U(0,30)$; specific conductance mean = $U(0,3000)$, SD $U(0,10,000)$; TOC mean = $U(0,100)$, SD = $U(0,500)$; pH mean = $U(0,10)$, SD = $U(0,15)$; SO_4 mean = $U(0,1500)$, SD = $U(0,2000)$; Ca mean = $U(0,50)$, SD = $U(0,75)$; Mg, K, and Na mean = $U(0,20)$, SD = $U(0,30)$). We used the R add-in library R2OpenBUGS (Sturtz et al., 2005), to organize our data into program R (2.14.0) (R Development Core Team, 2010), and used Markov chain Monte Carlo methods as implemented in OpenBUGS (Lunn et al., 2009) with three chains of 20,000 iterations, thinning factor of 1 after 5000 burn-in iterations to analyze each model. We evaluated the Markov chains by examining the history plots and the Gelman–Rubin statistic for each parameter for evidence of lack of convergence (Gelman and Rubin, 1992). The Gelman–Rubin statistic compares between- and within-chain variability; values near 1 (and up to 1.1) indicate likely convergence (Gelman and Hill, 2007).

We used a hierarchical Bayesian modeling approach to estimate species-specific and community responses to MTR/VF mining. This multi-level approach provided estimates of site-specific species richness in addition to separate estimates for species-specific occupancy and detection probabilities; therefore community-level and species-level attributes are incorporated into the same modeling framework (Dorazio and Royle, 2005; Zipkin et al., 2009). Specifically, we used a model similar to that used by Zipkin et al. (2009) and Hunt et al. (2013), to estimate species' occupancy and community responses to one site covariate (i.e., MTR/VF) and four survey covariates (*water temperature*, *date of last precipitation*, *Julian date* and *Julian date*²). One level of our model assumed a "true" (but only partially observed) presence–absence matrix z_{ij} for species $i = 1, 2, \dots, N$ at site $j = 1, 2, \dots, J$ where $z_{ij} = 1$ if a species i was present at site j , and $z_{ij} = 0$ if the species was absent at site j . Because z_{ij} was uncertain, we specified a model for occurrence, that used a Bernoulli distribution, where $z_{ij} \sim \text{Bern}(\Psi_{ij})$, and Ψ_{ij} is the probability that species i occurs at site j .

We used the salamander data we collected to generate species-specific encounter matrices for four sampling occasions at each stream. Within each species-specific matrix, detection was represented as 1 and non-detected was represented as 0. Thus, the data provided a three dimensional matrix x_{ijk} for species i at site j for the k th sampling occasion. An additional level of our model specified that $x_{ijk} \sim \text{Bern}(\Theta_{ijk}z_{ij})$ where z_{ij} is the true occurrence matrix described above, and the Θ_{ijk} is the detection probability for a species i at site j for the k th sampling occasion. This fulfills the condition that $x_{ijk} = 0$ if the species i is not present at site j , because in that case $z_{ij} = 0$.

We used the following equations to relate species-specific covariate parameters (α and β values) and occupancy and detection probabilities (Ψ_{ij} and Θ_{ijk} , respectively) to the hierarchical models we described above:

$$\text{logit}(\Psi_{ij}) = u_i + \alpha 1_i \text{MTR/VF}_j$$

$$\begin{aligned} \text{logit}(\theta_{ijk}) = & v_i + \beta 1_i \text{Julian date}_{jk} + \beta 2_i \text{Julian date}_{jk}^2 \\ & + \beta 3_i \text{water temperature}_{jk} \\ & + \beta 4_i \text{date of last precipitation}_{jk} \end{aligned}$$

The MTR/VF covariate was defined by whether the stream site was MTR/VF (represented as 1) or a control (represented as 0). *Julian date*, *water temperature*, and *date of last precipitation*, were assumed to influence detection rate of stream salamanders based on previous studies (see Spotila, 1972; Orser and Shure, 1975; Connette et al., 2011). *Julian date* was defined as the standardized score of Julian days since January 1, and *Julian date*² was defined as the squared standardized score of Julian days since January 1, *Water temperature* was defined as the standardized value of water temperature in degrees, and *date of last precipitation* was defined as the number of days since the last precipitation event. We included the Julian date (for linear effect) and Julian date squared (for squared effects along a normal distribution) because the capture probability, due to activity, may change during our sampling period from March to June. Standardization of covariates allowed for the estimation of Ψ and Θ at mean site and survey covariates from model-generated estimates of u_i (species-specific mean probability of occurrence) and v_i (species-specific mean probability of detection). Standardization of covariates also enabled direct comparison of the model coefficients as effect sizes relative to variation in each covariate. Our parameters u_i and v_i followed a joint normal distribution such that $[u_i, v_i | \Sigma] \sim N(0, \Sigma)$ (Dorazio et al., 2006), where Σ denotes a 2×2 symmetric matrix with diagonal elements σ^2_u and σ^2_v (the respective variances in u_i and v_i) and with off-diagonal elements σ_{uv} equal to the covariance in u_i and v_i (Dorazio and Royle, 2005).

Seven species-specific parameters were estimated by the model ($u_i, \alpha 1_i, v_i, \beta 1_i, \beta 2_i, \beta 3_i, \beta 4_i$). Community summaries (μ) were estimated by another hierarchical level of the model assuming that the species-specific parameters were random effects, each governed by a community-level hyper-parameter. For example, $\alpha 1_i \sim N(\mu_{\alpha 1}, \sigma_{\alpha 1})$ where $\mu_{\alpha 1}$ is the mean community response (across all species) to the MTR/VF covariate ($\alpha 1$), and $\sigma_{\alpha 1}$ is the standard deviation in $\alpha 1$ across species (Kéry et al., 2009). Using this hierarchical method, estimation of species-specific parameters can be precise, even where species are rare (Zipkin et al., 2009).

Our model used uninformative priors for the hyper-parameters and community summaries (e.g., $U(0,5)$ for all σ parameters and $U(-10$ to $10)$ for μ_{α} and μ_{β} parameters). We organized our data into program R (2.14.0) (R Development Core Team, 2010) and used the R add-in library R2OpenBUGS (Sturtz et al., 2005) to execute data analysis in the software program OpenBUGS (Lunn et al., 2009). Posterior summaries were based on 300,000 Markov chain Monte Carlo (MCMC) iterations, in which we disregarded the first 30,000 as burn-in with a thinning rate of 3. The mean and standard deviation of the model coefficients were calculated, in addition to the 2.5 and 97.5 percentiles of the distribution, which represent 95% Bayesian credible intervals. We used the log transformation (i.e., $(\exp(\alpha)/(1 + \exp(\alpha)))$ to derive species-specific occupancy and detection estimates. Convergence of the Markov chains was evaluated by observing the history plots and the Gelman–Rubin statistic (Gelman and Rubin, 1992). Lastly, with our model, we calculated mean species richness at MTR/VF sites and control sites, then calculated the pair-wise difference between mean species richness of MTR/VF sites and control sites and used 95% credible intervals to assess that difference.

3. Results

The average catchment size for control sites was 24.70 ha (± 21.34 SD), MTR/VF site average was 24.51 ha (± 15.48 SD).

Proportion of forest cover within the stream catchments and number of cover objects within the streams were greater at control streams than MTR/VF streams (Table 1). However, average wetted width (cm), and depth (cm) were similar between reference and MTR/VF stream transects (Table 1). Water chemistry attributes were consistently different between MTR/VF streams and control streams (Table 1). In particular, mean specific conductance was nearly 30 times greater at MTR/VF streams than at control sites and mean sulfate concentration was over 70 times greater at MTR/VF streams (Table 1). The remaining stream water quality attributes (temperature, pH, total organic carbon, Ca, Mg, K, Na) also were greater at MTR/VF stream compared to control streams (Table 1). For all environmental attributes, stationary distribution appeared to be achieved based on well-mixed history plots and the Gelman and Rubin statistic (<1.001 for all monitored parameters; Gelman and Rubin, 1992).

We detected 9 salamander species during our active searches; raw counts of salamander species at control sampling transects ranged from 2 to 6, species counts at MTR/VF sampling transects ranged from 0 to 4. However, we only considered 5 species (i.e., *Desmognathus fuscus*, *D. monticola*, *Gyrinophilus porphyriticus*, *Pseudotriton ruber* and *Eurycea cirrigera*) in our analysis as these species are primarily associated with streams. We detected a total of 97 salamanders at MTR/VF sites and 804 salamanders at control sites. Some species were rarely detected at MTR/VF sites; for example, only two *G. porphyriticus* and five *P. ruber* individuals were detected at MTR/VF streams. Mean baseline species detection probabilities were 0.57 (95% CI 0.41 to 0.71) for *D. fuscus*, 0.66 (95% CI 0.52 to 0.79) for *D. monticola*, 0.38 (95% CI 0.16 to 0.63) for *P. ruber*, 0.72 (95% CI 0.58 to 0.85) for *G. porphyriticus* and 0.67 (95% CI 0.54 to 0.79) for *E. cirrigera*. Model estimated detection parameters were not strongly influenced by sampling covariates.

Our model indicated high rates of mean species occupancy across all 5 species at control streams; mean estimated occupancy probabilities ranged from 0.73 (95% CI 0.41 to 0.96) for *P. ruber* to 0.90 (95% CI 0.77 to 0.98) for *E. cirrigera* (Fig. 1). Occupancy was lower at MTR/VF streams, with mean estimated occupancy probability ranging from 0.23 (95% CI 0.04 to 0.51) for *G. porphyriticus*, to 0.62 (95% CI 0.36 to 0.86) for *E. cirrigera* (Fig. 1). Despite having high posterior standard errors, we found that the species-specific α_{1i} parameter estimates were all negative and 95% credible intervals did not overlap with zero in any case, which collectively indicates that all species were less likely to occupy MTR/VF streams (i.e., *D. fuscus* $\alpha_{1i} = -1.78$ (95% CI -3.41 to -0.11), *D. monticola* $\alpha_{1i} = -2.07$ (95% CI -3.70 to -0.51), *P. ruber* $\alpha_{1i} = -2.38$ (95% CI -4.35 to -0.67), *G. porphyriticus* $\alpha_{1i} = -3.57$ (95% CI -6.16 to -1.69), *E. cirrigera* $\alpha_{1i} = -1.85$ (95% CI -3.46 to -0.23)). For our

model, stationary distributions appeared to be achieved based on well-mixed history plots and the Gelman and Rubin statistic (<1.001 for all monitored parameters; Gelman and Rubin, 1992).

When all the salamander species were considered together, as a community, the mean occupancy in MTR/VF streams was 0.50 (95% CI 0.06 to 0.95) and mean occupancy in control streams was 0.87 (95% CI 0.64 to 0.96) suggesting that salamanders have a higher probability of occupancy in streams that have not been affected by MTR/VF. The 95% credible interval for the occupancy covariate ($\mu_{\alpha 1}$ MTR/VF) contained only negative values -1.94 (95% CI -3.31 to -0.31) and the 95% credible interval for the standard deviation (i.e., 1.20 (95% CI 0.09 to 3.15), in the response to the covariate across species ($\sigma_{\alpha 1}$ MTR/VF) was less than the absolute value of the mean estimate, indicating certainty in the mean response across species (Table 2). All of the mean parameter estimates for detection covariates ($\mu_{\beta 1}$ – Julian date, $\mu_{\beta 2}$ – Julian date squared, $\mu_{\beta 3}$ – Water temperature, and $\mu_{\beta 4}$ – Date of last precipitation) covered zero and contained both positive and negative values in the 95% credible intervals, indicating uncertainty in the mean community responses to these covariates (Table 2). The mean species richness estimate for MTR/VF streams was 2.27 (± 1.27 SD) whereas richness was 4.67 (± 0.65 SD) for control streams (mean difference of 2.29 [95% CI 1.97 to 2.65]) between control and MTR/VF).

4. Discussion

We found that streams impacted by MTR/VF had reduced salamander species' occupancy and richness and altered environmental attributes compared to control streams. Recent research in West Virginia found that stream salamander abundance was reduced in first and second-order MTR/VF streams compared to reference streams, yet species richness did not differ between MTR/VF streams and control streams (Wood and Williams, 2013a,b). Based on our analysis, mean occupancy rates across five stream salamander species were reduced in MTR/VF compared to control streams. We recognize, however, that local abundance might be a source of systematic, detection bias in our study (see Royle and Nichols, 2003). If detectability is dependent on abundance then our model will not be able to separate sites with very low abundance from those unoccupied by salamanders. However, given the assumption of equal detectability between site types in our current analysis we conclude that occupancy and species richness differ between stream types.

Reduced salamander occupancy and species richness may be due to a complex set of interacting factors operating in both

Table 1
Mean, 95% credible intervals (95% CI), and differences in environmental attributes at mountaintop removal and valley fill (MTR/VF) and control (i.e., forest) streams located in the interior rugged section of the Cumberland Plateau in Breathitt and Knott Counties, Kentucky USA.

Variable	MTR/VF		Control		Difference	95% CI
	Mean	95% CI	Mean	95% CI		
Temperature (°C)	13.44	12.66 to 14.22	12.48	11.87 to 13.10	0.95	-0.03 to -1.95
Forest cover (%)	0.25	0.12 to 0.38	0.997	0.99 to 1.00	-0.75	-0.88 to -0.62
Specific Conductance ($\mu\text{S}/\text{cm}$)	1477.00	1103.00 to 1855.00	50.85	38.91 to 62.67	1427.00	1052.00 to 1804.00
Average stream width (cm)	122.60	88.33 to 156.70	130.60	102.3 to 159.1	-8.06	-52.47 to 35.54
Average stream depth (cm)	7.45	5.97 to 8.93	6.76	5.17 to 8.34	0.70	-1.47 to 2.85
Cover objects (#)	24.79	13.92 to 35.49	48.24	35.94 to 60.23	-23.45	-39.49 to -7.25
Total organic carbon (mg/l)	7.97	2.63 to 13.47	2.76	1.86 to 3.66	5.20	-0.21 to 10.77
pH (H ⁺)	6.08	5.35 to 6.82	5.71	5.34 to 6.09	0.37	-0.45 to 1.18
SO ₄ (mg/l)	506.70	260.10 to 758.20	7.22	3.47 to 10.99	499.50	252.90 to 751.30
Ca (mg/l)	23.72	21.79 to 25.65	1.28	1.10 to 1.45	22.44	20.51 to 24.38
Mg (mg/l)	10.14	9.75 to 10.54	1.62	1.40 to 1.83	8.53	8.08 to 8.97
K (mg/l)	8.15	6.04 to 10.26	2.11	1.08 to 3.13	6.04	3.72 to 8.40
Na (mg/l)	8.46	6.34 to 10.61	2.55	0.87 to 4.28	5.92	3.20 to 8.63

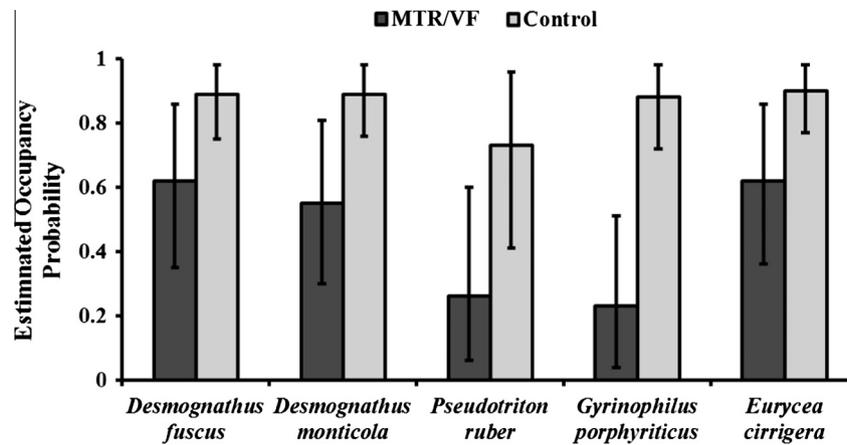


Fig. 1. Mean estimated occupancy probabilities (with 95% credible intervals) of stream salamanders detected in 10 m sampling transects at streams impacted by mountaintop removal and valley fill (MTR/VF) and streams within second growth forest (control). Study sites were located in the interior rugged section of the Cumberland Plateau, Kentucky, USA.

Table 2

Summary of hyper-parameters for occupancy and detection covariates for salamanders observed at sites of mountaintop removal and valley fill (MTR/VF) and natural second growth forest streams (controls) located in the interior rugged section of the Cumberland Plateau, Kentucky.

Community level hyper-parameter		Mean	Standard deviation	95% Credible interval	
^a $\mu_{\alpha 1}$	MTR/VF	-1.94	0.75	-3.31	-0.31
^a $\sigma_{\alpha 1}$	MTR/VF	1.20	0.80	0.09	3.15
$\mu_{\beta 1}$	Julian date	-0.05	0.21	-0.47	0.38
$\sigma_{\beta 1}$	Julian date	0.30	0.26	0.01	0.96
$\mu_{\beta 2}$	Julian date squared	0.08	0.20	-0.32	0.49
$\sigma_{\beta 2}$	Julian date squared	0.29	0.25	0.01	0.94
$\mu_{\beta 3}$	Water temperature	0.13	0.20	-0.27	0.52
$\sigma_{\beta 3}$	Water temperature	0.23	0.22	0.01	0.81
$\mu_{\beta 4}$	Date of last precipitation	-0.07	0.20	-0.48	0.33
$\sigma_{\beta 4}$	Date of last precipitation	0.28	0.25	0.01	0.93

^a The symbol μ indicates mean community response, while σ indicates the standard deviation in the response to the covariate across species.

terrestrial and aquatic habitats. First, the deposition of overburden into valleys results in the permanent loss or burial of most of the length of low-order streams within valleys (Palmer et al., 2010). The permanent loss of streams likely reduces connectivity among salamander populations across landscapes, leading to reduced gene flow and possible local extinction for some species (i.e., Munshi-South et al., 2013). Second, MTR/VF streams often have reduced forest cover within catchments, which has been shown to be negatively correlated with salamander occupancy rates and abundances (i.e., Ford et al., 2002; Price et al., 2011, 2012). Indeed, the MTR/VF streams had, on average, 75% less forest cover than control streams; land-cover within MTR/VF catchments was dominated by non-native grasses and shrubs. Reduction of forest cover within stream catchments may be particularly detrimental to species such as *P. ruber* and *E. cirrigera*, which extensively use terrestrial habitats during the non-breeding season (Petranka, 1998). Additionally, Wood and Williams (2013b) noted lower terrestrial salamander abundance and species richness within reclaimed, grass-dominated surface mine and suggest that poor soils, reduced vertical structure of vegetation, little tree cover, and inadequate litter and wood debris cover contributed to their findings. Natural history differences, such as use of terrestrial habitat by stream salamanders, may contribute to the interspecific differences we observed in occupancy; however formal tests are needed to determine relationships between salamander natural-history traits and sensitivity to MTR/VF.

Land-cover changes on MTR/VF sites lead to numerous changes in hydrology and alterations to in-stream habitat, which may also lead to decreased salamander occupancy and species richness.

Reclaimed mine sites have soils containing unweathered rock that is heavily compacted to reduce erosion, resulting in altered water tables and disturbed flow paths (Bonta et al., 1992; Bernhardt and Palmer, 2011). In particular, compacted soils lead to high rates of storm water runoff. Negley and Eshleman (2006) and Ferrari et al. (2009) found that MTR/VF streams had tripled storm runoff and doubled flow rates compared to reference catchments. High peak flows have been shown to negatively affect survival of larval *E. cirrigera* in urban settings (Barrett et al., 2010) and may influence survival and occupancy within MTR/VF streams. Altered hydrology is often apparent through stream bank erosion and sedimentation, which can be excessive in MTR/VF streams (Fox, 2009). Sedimentation results in the burial of rocks and boulders and the infilling of interstitial spaces between rocks, which reduces available microhabitats for salamanders (Lowe et al., 2004). Cover objects were reduced in our MTR/VF streams, which may have been due to burial of rocks by sediment. Wood and Williams (2013a) suggest that sedimentation contributed to lower abundances of stream salamanders in West Virginia MTR/VF streams and Redmond (1980) found Black Mountain dusky salamanders (*Desmognathus walteri*) were excluded from highly silted streams due to coal mining.

We found MTR/VF streams had elevated levels of specific conductance, sulfates, total organic carbon, and dissolved ion concentrations. A previous study conducted at the Laurel Fork mine also found elevated specific conductance levels and dissolved ion concentrations at three of our study sites (Fritz et al., 2010), and numerous investigations on the effects of MTR/VF on water chemistry corroborate our results (i.e., Hartman et al., 2005; Pond et al., 2008; Wood and Williams, 2013a). Amphibians are poor osmoreg-

ulators; high specific conductance has been shown to have a wide range of adverse effects (i.e., physical abnormalities, reduced survivorship, reduced activity, increased corticosterone levels) on larval stages of amphibians (Sanzo and Hecnar, 2006; Karraker et al., 2008; Chambers, 2011), perhaps resulting in population declines and species extirpations. Miller et al. (2007) found that larval *E. cirrigera* abundance was negatively related to specific conductance levels in urban streams and Schorr et al. (2013) found that occurrences of four salamander species of the Cumberland Plateau (*D. fuscus*, *P. ruber*, *E. cirrigera*, *G. porphyriticus*) were negatively correlated with elevated specific conductance levels (i.e., >100 $\mu\text{S}/\text{cm}$). Stream invertebrates are an important prey item for salamanders (Petranka, 1998; Davic and Welsh, 2004) and decreases in macroinvertebrate populations due to water chemistry are well documented in streams impacted by MTR/VF (Pond, 2010,2012). Thus, adverse effects on larval amphibians combined with a reduction in prey items may lead to decreases in salamander occupancy and species richness.

The disturbance caused by MTR/VF is drastically changing the central Appalachian landscape, compromising the natural ecological and functional state of both terrestrial and aquatic environments. The reclamation process, emphasizing soil compaction and the establishment of non-native herbaceous species, has hindered the establishment of native tree species on MTR sites (Zipper et al., 2011). These terrestrial impacts in combination with changes in water chemistry and stream geomorphology lead to long-lasting changes to terrestrial and aquatic ecosystem function (Simmons et al., 2008). Full recovery of species diversity in streams impacted by MTR/VF has not been documented (Palmer et al., 2010). Because stream salamanders use both terrestrial and aquatic habitats; it is not surprising that we found that MTR/VF resulted in reduced occupancy and species richness. Although there is no evidence suggesting that chemical and hydrological alterations of streams by MTR/VF can be ameliorated by current reclamation procedures (Bernhardt and Palmer, 2011), the Forestry Reclamation Approach that advocates reforesting MTR/VF land, could be beneficial for salamanders (and other aquatic and semi-aquatic animals) via not only increasing forest cover within catchments, but also by influencing hydrology and water chemistry within the disturbed watershed (Burger et al., 2005; Zipper et al., 2011). However, research documenting the proximate mechanisms driving reduced salamander occupancy and species richness is likely needed if recovery is to be successful.

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References

- Barrett, K., Helms, B.S., Samoray, S.T., Guyer, C., 2010. Growth patterns of a stream vertebrate differ between urban and forested catchments. *Freshw. Biol.* 55, 1628–1635.
- Barton, C., 2011. Coal mining versus water quality: an electrifying topic. *Am. Water Resour. Assoc.: Water Resour. Impact* 13, 23–24.
- Bernhardt, E.S., Palmer, M.A., 2011. The environmental costs of mountaintop mining valley fill operations for aquatic ecosystems of the central Appalachians. *Year in ecology and conservation biology. Ann. N. Y. Acad. Sci.* 1223, 39–57.
- Bernhardt, E.S., Lutz, B.D., King, R.S., Fay, J.P., Carter, C.E., Helton, A.M., Campagna, D., Amos, J., 2012. How many mountains can we mine? Assessing the regional degradation of central Appalachian rivers by surface coal mining. *Environ. Sci. Technol.* 46, 8115–8122.
- Bonta, J.V., Amerman, C.R., Dick, W.A., Hall, G.F., Harlukowicz, T.J., Razem, A.C., Smeck, N.E., 1992. Impact of surface coal mining on three Ohio watersheds – physical conditions and ground-water hydrology. *Water Resour. Bull.* 28, 577–596.
- Bridge, G., 2004. Contested terrain: mining and the environment. *Annu. Rev. Environ. Resour.* 29, 205–259.
- Burger, J., Graves, D., Angel, P., Davis, V., Zipper, C., 2005. The Forestry Reclamation Approach. Appalachian Regional Reforestation Initiative, US Office of Surface Mining. Forest Reclamation Advisory Number 2.
- Chambers, D.L., 2011. Increased conductivity affects corticosterone levels and prey consumption in larval amphibians. *J. Herpetol.* 45, 219–223.
- Connette, G.M., Price, S.J., Dorcas, M.E., 2011. Influence of abiotic factors on activity in a larval salamander assemblage. *Southeast. Nat.* 10, 109–120.
- Cooke, J.A., Johnson, M.S., 2002. Ecological restoration of land with particular reference to the mining of metals and industrial minerals: a review of theory and practice. *Environ. Rev.* 10, 41–71.
- Davic, R.B., Welsh Jr., H.H., 2004. On the ecological roles of salamanders. *Annu. Rev. Ecol. Evol. Syst.* 35, 405–434.
- Dorazio, R.M., Royle, J.A., 2005. Estimating size and composition of biological communities by modeling the occurrence of species. *J. Am. Stat. Assoc.* 100, 389–398.
- Dorazio, R.M., Royle, J.A., Soderstrom, B., Glimskar, A., 2006. Estimating species richness and accumulation by modeling species occurrence and detectability. *Ecology* 87, 842–854.
- Ferrari, J.R., Lookingbill, T.R., McCormick, B., Townsend, P.A., Eshleman, K.N., 2009. Surface mining and reclamation effects on flood response of watersheds in the central Appalachian Plateau region. *Water Resour. Res.* 45, W04407. <http://dx.doi.org/10.1029/2008WR007109>.
- Ferrari, C.P., Stauffer, J.R., Stecko, T.D., 2004. Evaluating impacts of mountain top removal/valley fill coal mining on stream fish populations. In: 2004 National Meeting of the American Society of Mining and Reclamation, pp. 576–592.
- Ford, W.M., Chapman, B.R., Menzel, M.A., Odum, R.H., 2002. Stand age and habitat influences on salamanders in Appalachian cove hardwood forests. *For. Ecol. Manage.* 155, 131–141.
- Fox, J.F., 2009. Identification of sediment sources in forested watersheds with surface coal mining disturbance using carbon and nitrogen isotopes. *J. Am. Water Resour. Assoc.* 45, 1273–1289.
- Fritz, K.M., Fulton, S., Johnson, B.R., Barton, C.D., Jack, J.D., Word, D.A., Burke, R.A., 2010. Structural and functional characteristics of natural and constructed channels draining a reclaimed mountaintop removal and valley fill coal mine. *J. North Am. Benthol. Soc.* 29, 673–689.
- Gelman, A., Hill, J., 2007. *Data Analysis Using Regression and Multilevel/Hierarchical Models*. Cambridge University Press, Cambridge.
- Gelman, A., Rubin, D.B., 1992. Inference from iterative simulation using multiple sequences. *Stat. Sci.* 7, 457–472.
- Grant, E.H.C., Green, L.E., Lowe, W.H., 2009. Salamander occupancy in headwater stream networks. *Freshw. Biol.* 54, 1370–1378.
- Greenberg, A.E., Clesceri, L.S., Eaton, A.D., 1992. *Standard Methods for the Examination of Water and Wastewater*, 18th ed. American Public Health Association, Washington, DC.
- Griffith, M.B., Norton, S.B., Alexander, L.C., Pollard, A.I., LeDuc, S.D., 2012. The effects of mountaintop mines and valley fills on the physicochemical quality of stream ecosystems in the central Appalachians: a review. *Sci. Total Environ.* 417 (418), 1–12.
- Hartman, K., Kaller, M., Howell, J., Sweka, J., 2005. How much do valley fills influence headwater streams? *Hydrobiologia* 532, 91–102.
- Hunt, S.D., Guzy, J.C., Price, S.J., Halstead, B.J., Eskew, E.A., Dorcas, M.E., 2013. Responses of riparian reptile communities to damming and urbanization. *Biol. Conserv.* 157, 277–284.
- Karraker, N.E., Gibbs, J.P., Vonesh, J.R., 2008. Impacts of road deicing salt on the demography of vernal pool-breeding amphibians. *Ecol. Appl.* 18, 724–734.
- Keitzer, S.C., Goforth, R.R., 2013. Salamander diversity alters stream macroinvertebrate community structure. *Freshw. Biol.* 58, 2114–2125.
- Kéry, M., 2010. *Introduction to WinBUGS for Ecologists: A Bayesian Approach to Regression, ANOVA, Mixed Models and Related Analyses*. Academic Press, USA.
- Kéry, M., Royle, J.A., Plattner, M., Dorazio, R.M., 2009. Species richness and occupancy estimation in communities subject to temporary emigration. *Ecology* 90, 1279–1290.
- Lindberg, T.T., Bernhardt, E.S., Bier, R., Helton, A.M., Merola, R.B., Wengosh, A., Di Giulio, R.T., 2011. Cumulative impacts of mountaintop mining on an Appalachian watershed. *Proc. Natl. Acad. Sci. USA* 108, 20929–20934.
- Litz, B.D., Bernhardt, E.S., Schlesinger, W.H., 2013. The environmental price tag on a ton of mountaintop removal coal. *PLoS One* 8, e73203.
- Lowe, W.H., Nislow, K.H., Bolger, D.T., 2004. Stage-specific and interactive effects of sedimentation and trout on a headwater stream salamander. *Ecol. Appl.* 14, 164–172.
- Lunn, D., Spiegelhalter, D., Thomas, A., Best, N., 2009. The BUGS project: evolution, critique, and future directions. *Stat. Med.* 28, 3049–3067.
- Miller, J.E., Hess, G.R., Moorman, C.E., 2007. Southern two-lined salamanders in urbanizing watersheds. *Urban Ecosyst.* 10, 73–85.

- Munshi-South, J., Zak, Y., Pehek, E., 2013. Conservation genetics of extremely isolated urban populations of the northern dusky salamander (*Desmognathus fuscus*) in New York City. *PeerJ* 1, e64.
- Negley, T.L., Eshleman, K.N., 2006. Comparison of storm-flow responses of surface-mined and forested watersheds in the Appalachian Mountains, U.S.A. *Hydrol. Process.* 20, 3467–3483.
- Orser, P.N., Shure, D.J., 1975. Population cycles and activity patterns of the Dusky salamander, *Desmognathus fuscus fuscus*. *Am. Midl. Nat.* 93, 403–410.
- Palmer, M.A., Bernhardt, E.S., Schlesinger, W.H., Eshleman, K.N., Foufoula-Georgiou, E., Hendryx, M.S., Lemly, A.D., Likens, G.E., Loucks, O.L., Power, M.E., White, P.S., Wilcock, P.R., 2010. Mountaintop mining consequences. *Science* 327, 148–149.
- Petranka, J.W., 1998. Salamanders of the United States and Canada. Smithsonian Institution Press, Washington, DC.
- Phillippi, M.A., Boebinger, A., 1986. A vegetational analysis of three small watersheds in Robinson Forest, Eastern Kentucky. *Castanea* 51, 11–30.
- Pond, G.J., 2010. Patterns of Ephemeroptera taxa loss in Appalachian headwater streams (Kentucky, USA). *Hydrobiologia* 641, 185–201.
- Pond, G.J., 2012. Biodiversity loss in Appalachian headwater streams (Kentucky, USA): Plecoptera and Trichoptera communities. *Hydrobiologia* 679, 97–117.
- Pond, G.J., Passmore, M.E., Borsuk, F.A., Reynolds, L., Rose, C.J., 2008. Downstream effects of mountain top coal mining: comparing biological conditions using family- and genus-level macroinvertebrate bioassessment tools. *J. Am. Benthol. Soc.* 27, 717–737.
- Price, S.J., Cecala, K.K., Browne, R.A., Dorcas, M.E., 2011. Effects of urbanization on occupancy of stream salamanders. *Conserv. Biol.* 25, 547–555.
- Price, S.J., Browne, R.A., Dorcas, M.E., 2012. Evaluating the effects of urbanization on salamander abundances using a before–after control–impact design. *Freshw. Biol.* 57, 193–203.
- R Development Core Team, 2010. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.
- Redmond, W.H., 1980. Notes on the distribution and ecology of the black mountain dusky salamander *Desmognathus walteri* Barbour (Amphibia: Plethodontidae) in Tennessee. *Brimleyana* 4, 123–131.
- Royle, J.A., Nichols, J.D., 2003. Estimating abundance from repeated presence–absence data or point counts. *Ecology* 84, 777–790.
- Sanzo, D., Hecnar, S.J., 2006. Effects of road de-icing salt (NaCl) on larval wood frogs (*Rana sylvatica*). *Environ. Pollut.* 140, 247–256.
- Schorr, M.S., Dyson, M.C., Nelson, C.H., Van Horn, G.S., Collins, D.E., Richards, S.M., 2013. Effects of stream acidification on lotic salamander assemblages in a coal-mined watershed in the Cumberland Plateau. *J. Freshw. Ecol.* 28, 339–353.
- Simmons, J.A., Currie, W.S., Eshleman, K.N., Kuers, K., Monteleone, S., Negley, J.L., Pohlard, B.R., Thomas, C.L., 2008. Forest to reclaimed mine land use change leads to altered ecosystem structure and function. *Ecol. Appl.* 18, 104–118.
- Spotila, J.R., 1972. Role of temperature and water in the ecology of lungless salamanders. *Ecol. Monogr.* 42, 95–125.
- Sturtz, S., Ligges, U., Gelman, A., 2005. R2WinBUGS: a package for running WinBUGS from R. *J. Stat. Softw.* 12, 1–16.
- Warren, M.L., Haag, W.R., 2005. Spatio-temporal patterns of the decline of freshwater mussels in the little South Fork Cumberland River, USA. *Biodivers. Conserv.* 14, 1383–1400.
- Wood, P.B., Williams, J.M., 2013a. Impact of valley fills on streamside salamanders in Southern West Virginia. *J. Herpetol.* 47, 119–125.
- Wood, P.B., Williams, J.M., 2013b. Terrestrial salamander abundance on reclaimed mountaintop removal mines. *Wildl. Soc. Bull.* 37, 815–823.
- Zipkin, E.F., DeWan, A., Royle, J.A., 2009. Impacts of forest fragmentation on species richness: a hierarchical approach to community modelling. *J. Appl. Ecol.* 46, 815–822.
- Zipper, C.E., Burger, J.A., Skousen, J.G., Angel, P.N., Barton, C.D., Davis, V., Franklin, J.A., 2011. Restoring forests and associated ecosystem services on Appalachian coal surface mines. *Environ. Manage.* 47, 751–765.