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THE EFFECTS OF MOUNTAINTOP REMOVAL MINING AND VALLEY FILLS ON STREAM SALAMANDER COMMUNITIES

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THESIS

A thesis submitted in partial fulfillment of the
requirements for the degree of Master of Science in the
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By
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ABSTRACT OF THESIS

THE EFFECTS OF MOUNTAINTOP REMOVAL MINING AND VALLEY FILLS ON STREAM SALAMANDER COMMUNITIES

Mountaintop removal mining and valley filling (MTR/VF) is a common form of land conversion in Central Appalachia and threatens the integrity of stream ecosystems. We investigated the effects of MTR/VF on stream salamander occupancy probabilities and community structure by conducting area constrained active searches for stream salamanders within intermittent streams located in mature forest (i.e., control) and those impacted by MTR/VF. During March to June of 2013, we detected five stream salamander species (*Desmognathus fuscus*, *D. monticol, Eurycea cirrigera*, *Pseudotriton ruber*, and *Gyrinophilus porphyriticus*) and found that the probability of occupancy was greatly reduced in MTR/VF streams compared to control streams. Additionally, the salamander community was greatly reduced in MTR/VF streams; the mean species richness estimate for MTR/VF streams was 2.09 (± 1.30 SD), whereas richness was 4.83 (± 0.58 SD) for control streams. Numerous mechanisms may be responsible for decreased occupancy and diminished salamander communities at MTR/VF streams, although water chemistry of streams may be a particularly important mechanism. Indeed, we detected elevated levels of specific conductivity, pH, total organic carbon, and dissolved ions in MTR/VF streams. Our results indicate that salamander communities, with other invertebrates, fish, and other aquatic and/or semi-aquatic animals, are susceptible to MTR/VF mining practices.

KEYWORDS: amphibians, Appalachia, coal mining, hierarchical Bayesian modeling, occupancy

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5 March 2014

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CHAPTER 1: INTRODUCTION

Mountaintop removal has become the dominant type of mining for the extraction of shallow coal seams in Appalachia. 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 atmospheric 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 conductivity, and elevated levels of total dissolved solids (i.e., sulfates (SO_4), calcium, magnesium) 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 mountaintop removal mined lands, streams impacted by mountaintop removal mining and valley fill (MTR/VF) typically have altered hydrology (i.e., decreased infiltration, increased peak flows) compared to streams within forested catchments (Negley & Eshleman 2006). More than 1.1 million ha of forest land has been altered by surface mining in central Appalachia, USA (Bernhardt & Palmer 2011). In the Commonwealth of Kentucky, approximately 2,000 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).

Appalachian streams influenced by MTR/VF are often characterized by diminished biological communities in comparison to reference streams. For example, macroinvertebrate (including Ephemeroptera, Plecoptera, and Trichoptera taxa) communities in MTR/VF streams have significant reductions in species richness compared to reference locations (Pond 2010; Pond 2012), and decreases in freshwater mussel diversity is positively correlated with extent of surface mines within catchments of central Appalachian rivers (Warren & Haag 2005). Additionally, in Kentucky streams affected by MTR/VF, observed stream sites downstream of VF had a 50% reduction in fish species richness compared to streams without MTR/VF in the watershed (Ferreri et al. 2004). Amphibians, specifically salamanders, are important components of low-order stream ecosystems (Davic & Welsh 2004); up to 9 species can co-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 & Welsh 2004; Keitzer & Goforth 2013). Yet, investigations on the responses of stream salamander communities to MTR/VF mining are lacking.

To evaluate the effects of MTR/VF mining on stream salamander communities, we compared species occupancy and community composition of stream salamanders within stream catchments 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 despite 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 richness in salamander communities, and that MTR/VF streams would differ significantly in water chemistry and habitat characteristics from control locations.

CHAPTER 2: METHODS

Study Sites –

We investigated salamander communities at 23 intermittent 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 streams located on the reclaimed Laurel Fork surface mine and 12 control streams in natural, second growth forest on the University of Kentucky's Robinson Forest, which is located directly northeast of the Laurel Fork surface mine (See Figure 2.1 for map, Table 2.1 for coordinates). Control sites were typical, mixed mesophytic forests of the region; dominant tree species on lower catchment slopes consisted of white oak (*Quercus alba*), tulip tree (*Liriodendron tulipifera*), Eastern hemlock (*Tsuga Canadensis*), and American beech (*Fagus grandifolia*). Black oak (*Quercus velutina*) and chestnut oak (*Quercus prinus*) were dominant on the upper, more xeric slopes (Phillippi & Boebinger 1986).

Our MTR/VF study stream sites at Laurel Fork were surface mined in the late 1990s to the early 2000s and have perimeter drains. The dominant vegetation cover of the MTR/VF catchments included the nitrogen-fixing herb Sericea lespedeza (*Lespedeza cuneata*) and grasses, primarily tall fescue (*Schedonorus arundinaceus*), with autumn olive (*Elaeagnus umbellate*) shrubs scattered throughout the landscape. Young stands are scattered on the Laurel Fork mined land including, white pine (*Pinus strobus*), Virginia pine (*Pinus virginiana*), white oak (*Q. alba*), American sycamore (*Platanus occidentalis*) and black locust (*Robinia pseudoacacia*). See Fritz et al. (2010) for additional information on the Laurel Fork study site.

Data Collection Methods –

Area-constrained active searches were used to sample salamanders at each site, in a 10-m stream transect. Transects were chosen on the basis of their similarity of width, depth and approximated current velocity, as well as including a riffle, run, and pool where posssible. MTR/VF streams were sampled below the valley fill. We used a combination of systematic dipnetting and bank searches to capture salamanders (Willson & Dorcas 2003; Price et al. 2012). 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 stream. One person also conducted bank searches, which included searching under rocks, logs, leaf litter and other material within 1 m of the bank. In general, dipnetting sessions took approximately 30 minutes and bank searches took 15 minutes to complete. 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 transect was sampled four times (i.e., usually monthly) from March through June 2013 and all searches were conducted during daylight hours.

We recorded several variables before each active search. Prior to sampling, we measured stream width and depth at the start, middle, and end of each 10 m sampling transects and counted the number of cover objects, which included rocks approximately 15 cm or larger in diameter and logs with approximately 8 cm or larger in diameter, within our sampling transects. Also prior to sampling, 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 transported to the University of Kentucky Forestry Hydrology Laboratory where they were analyzed for concentrations of calcium (Ca), magnesium (Mg), potassium (K), sodium (Na), sulfates (SO_4^{2-}), and total organic carbon (TOC), pH and specific conductivity. Water sampling, preservation, and analytic protocols were performed in accordance with standard methods (Greenberg et al. 1992). Concentrations of SO_4^{2-} were determined by means of a quantitative ion chromatography procedure on an Ion Chromatograph 2000 (Dionex Corporation, Sunnyvale, California). Measurements of Ca, Mg, K, and Na concentrations were made with a GBC SDS 270 Atomic Absorption Spectrophotometer (GBC Scientific Equipment, Melbourne, Australia). Total organic carbon was analyzed on acidified samples with Shimadzu TOC-5000A (Shimadzu America, Columbia, Maryland). Specific conductivity was measured using a YSI conductivity bridge (YSI, Yellow Springs, Ohio) and pH was measured using an Orion pH-meter (Thermo Fisher Scientific, Waltham, Massachusetts). Finally, we used a geographic information system (ArcGIS 10.1 ESRI) and Watershed tool in ArcToolBox, to calculate the catchment area and land-cover composition of each of our study sites.

Data Analysis –

We compared several environmental attributes between control streams and MTR/VF streams using Bayesian analysis t-tests with unequal variances (Kėry 2010). Attributes included: forest cover, average stream width and depth in our sampling transects, number of cover objects within our sampling transects, water temperature, specific conductivity, 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., forest cover mean = 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 conductivity mean = U(0, 3000), SD U(0, 10000); 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 organized our data into program R (2.14.0) (R Development Core Team 2010) and, using the R add-in library R2WinBUGS (Sturtz et al. 2005), analyzed each model using Markov chain Monte Carlo methods as implemented in WinBUGS (Lunn et al. 2000) with three chains of 20,000 iterations, thinning factor of 1 after 5,000 burn-in iterations. We evaluated the models by examining the history plots and the Gelman-Rubin statistic for each parameter for evidence of lack of convergence (Gelman & Rubin 1992).

We used a hierarchical Bayesian modeling approach to estimate salamander community and species-specific responses to MTR/VF mining. This multi-level approach provided estimates of site-specific species richness in addition to separate estimates for species-specific occurrence and detection probabilities; therefore community-level and species-level attributes are incorporated into the same modeling framework (Dorazio & Royle 2005; Zipkin et al. 2009). Specifically, we employed a similar model to that used by Zipkin et al. (2009) and Hunt et al. (2013) to estimate species 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 $i = 1, 2,\ldots, N$ at site $j = 1, 2,\ldots, J$ 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, using a Bernoulli distribution, where $z_{ij} \sim Bern(\Psi_{ij})$, and Ψ_{ij} is the probability that a species i occurs at site j.

Using the salamander data we collected, species-specific encounter matrices were generated for four sampling occasions at each stream (See Appendix A for species encounter matrices). 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 kth sampling occasion. An additional level of our model specified that $x_{ijk} \sim Bern\ (\Theta_{ijk} z_{ij})$ 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 kth 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:

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

$$logit(\theta_{ijk}) = v_i + \beta 1_i Juliandate_{jk} + \beta 2_i Juliandate^2_{jk} + \beta 3_i watertemperature_{jk} + \beta 4_i Dateoflastprecipitation_{jk}$$

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* 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. These covariates, Julian date, water temperature, and number of days since last precipitation event, were assumed to influence detection rate of stream salamanders. Water temperature and number of days since last precipitation have been suggested as important predictors of activity in stream salamanders (Spotila 1972; Orser & Shure 1975; Connette et al. 2011). We included the Julian date (for linear effect) and Julian date squared (for squared effects along a normal distribution) because the capture events, 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 $[u_i, v_i \mid \Sigma] \sim N(0, \Sigma)$ (Dorazio et al. 2006), where Σ denotes a 2 x 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 & Royle 2005).

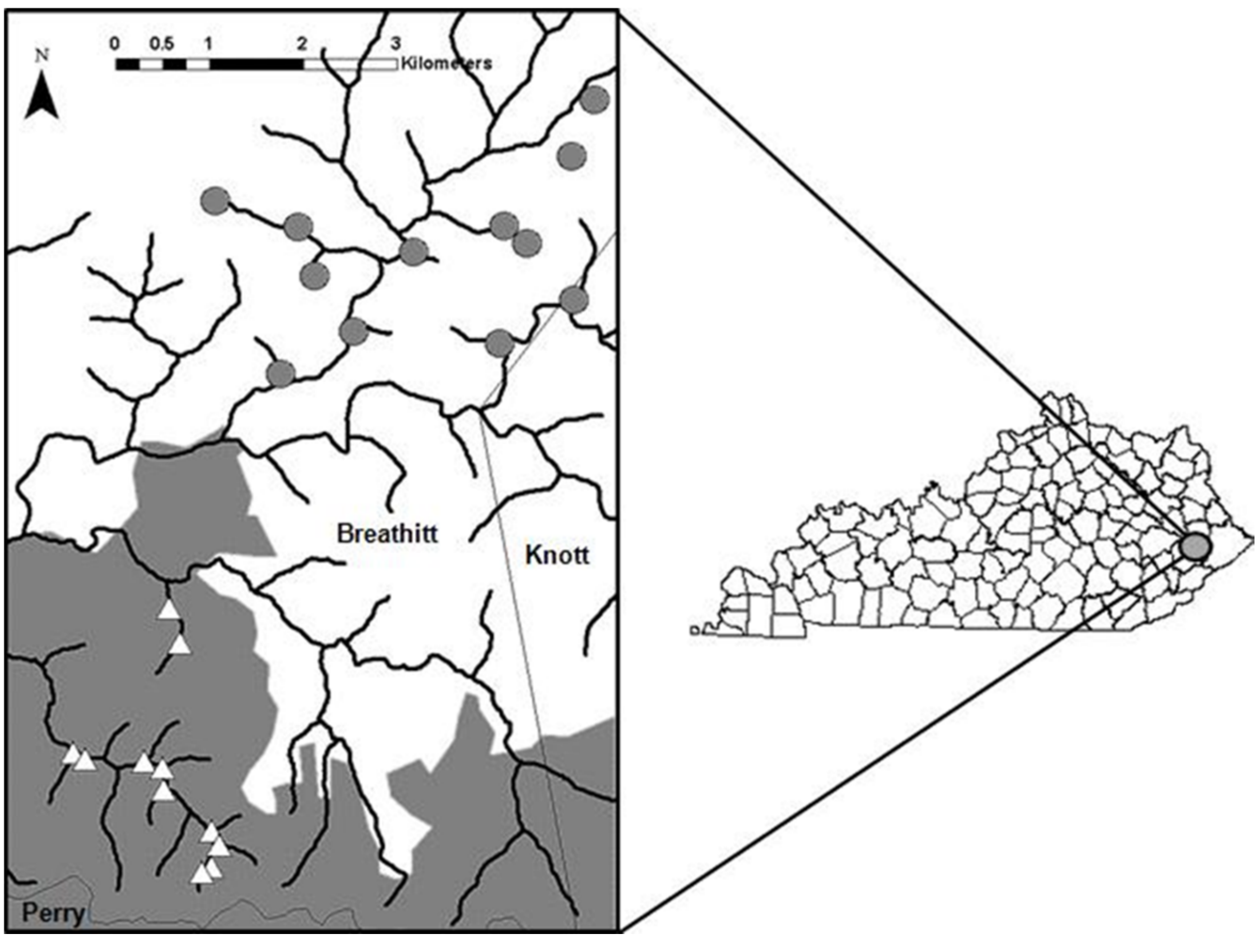
Seven species-specific parameters were included in the model (u_i, $\alpha1_i$, v_i, $\beta1_i$, $\beta2_i$, $\beta3_i$, $\beta4_i$). Community summaries (μ) were estimated by another hierarchical level of the model assuming that the species-specific parameters were random effects, each governed by a community-level hyper-parameter. For example, $\alpha1_i \sim N(\mu_{\alpha1}, \sigma_{\alpha1})$ where $\mu_{\alpha1}$ is the mean community response (across all species) to the *MTR/VF* covariate ($\alpha1$), and $\sigma_{\alpha1}$ is the standard deviation in $\alpha 1$ across species (Kèry et al. 2009). Using this hierarchical method, estimation of species-specific parameters is allowed, 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 $U(-10 \text{ to } 10)$ for μ_α and μ_β parameters). We organized our data into program R (2.14.0) (R Development Core Team 2010) and executed data analysis in the software program OpenBUGS (Lunn et al. 2000) using the R add-in library R2OpenBUGS (Sturz et al. 2005) (See Appendix B.01. for R code used for occupancy and species richness analysis). 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. Species-specific occupancy and detection estimates were derived using the log transformation (i.e., $(\exp(\alpha)/(1 + \exp \alpha))$). The models were evaluated by observing the history plots and the Gelman-Rubin statistic (Gelman & Rubin 1992). Lastly, using 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 assessed that difference using 95% credible intervals and standard deviations (See Appendix B.02. for R code for estimated average species richness across all sites).

Table 2.1. Coordinates and catchment sizes for sample stream sites at control mature forest streams and MTR/VF streams located in Breathitt and Knott counties, Kentucky on the Cumberland Plateau of Appalachia.

Site	MTR/VF or Control	Catchment size (ha)	Easting	Northing	Zone
Miller Hollow	Control	7.98	312312.88	4150387.60	17
Goff Hollow	Control	19.14	312590.69	4150985.30	17
Falling Rock A	Control	12.06	311561.99	4149683.55	17
Falling Rock B	Control	17.47	311796.94	4149486.70	17
Little Millseat A	Control	14.99	309364.88	4149785.15	17
Little Millseat B	Control	10.53	308494.93	4150109.00	17
Boardinghouse	Control	31.13	309104.35	4148223.04	17
Coles A	Control	87.27	312266.05	4148858.84	17
Bucklick Hollow	Control	15.64	311452.45	4148428.89	17
A Field	Control	17.28	309507.36	4149256.77	17
Mulberry	Control	32.90	309896.07	4148643.56	17
Tome	Control	30.08	310585.46	4149449.73	17
Spicewood	MTR/VF	28.95	306692.85	4144299.79	17
Turkey	MTR/VF	6.89	306811.91	4144220.42	17
Wharton	MTR/VF	61.53	307440.30	4144167.50	17
Hickory Log	MTR/VF	13.88	307635.43	4144091.43	17
Big Hollow	MTR/VF	18.74	307638.74	4143856.61	17
Stillrock	MTR/VF	12.69	308121.60	4143383.67	17
White Oak	MTR/VF	24.50	308191.06	4143234.84	17
Unnamed R White Oak	MTR/VF	32.03	308105.07	4143003.33	17
Unnamed L White Oak	MTR/VF	10.81	307989.31	4142947.11	17
Bee Far	MTR/VF	22.47	307788.23	4145786.29	17
Bee Near	MTR/VF	37.17	307883.87	4145413.46	17

Figure 2.1. Active search study sites at 23 intermittent streams located in the interior rugged section of the Cumberland Plateau in Breathitt and Knott counties, Kentucky USA. Eleven streams were located on the reclaimed Laurel Fork surface mine (4144091.438 N, 307635.435 E; mined area in filled grey) (white triangles) and 12 streams were located in natural second growth forest on the University of Kentucky's Robinson Forest, adjacent to Laurel Fork surface mine (grey circles).



CHAPTER 3: RESULTS

The average catchment size for control sites was 24.70±21.34 ha (SD), MTR/VF site average was 24.51±15.47 ha (SD). 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 & Rubin 1992). Proportion of forest cover within the stream catchments was greater at control streams (mean = 0.997 (95% CI 0.993-0.999), than MTR/VF streams (mean = 0.25 (95% CI 0.12-0.38) (difference = -0.75 [95% CI -0.88-(-0.62)]; Table 3.1)). Average stream width (cm), depth (cm) and number of cover objects were similar between reference and MTR/VF stream transects (differences = width -8.06 [95% CI -52.47-35.54], depth 0.70 [95% CI -1.47-2.85], cover objects -23.45 [95% CI -39.49-(-7.25)]). Water chemistry attributes were consistently different between MTR/VF streams and control streams (Table 3.1). In particular, mean specific conductivity 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 3.1). 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 3.1).

We detected 9 salamander species during our active searches; raw counts of salamander species at control streams ranged from 2 to 6, species counts at MTR/VF streams ranged from 0 to 4. However, we only considered 5 species (i.e., *D. fuscus*, *D. monticola*, *E. cirrigera*, *G. porphyriticus*, and *P. ruber*) in our analysis as these species are primarily associated with streams (Figure 3.1). *Plethodon* species, *Hemidactylium scutatum*, and *Notophthalamus viridscens* were detected in riparian areas, but are not obligatorily tied to streams from reproduction and were removed from analysis. We detected a total of 97 individual salamanders at MTR/VF sites and 804 individuals at control sites. Some species were rarely detected at MTR/VF sites; for example, only two *G. porphyritcus* and five *P.ruber* were detected at MTR/VF streams. Mean estimated species detection probabilities ranged from 0.36 (95% CI 0.15-0.62) for *P. ruber* to 0.72 (95% CI 0.58-0.86) for *G. porphyriticus* (Figure 3.1; See Appendix C.01 for mean estimated detection probabilities). Model estimated detection parameters were not strongly influenced by sampling covariates (Table 3.2).

Our model indicated high rates of mean species occupancy across all 5 species at control streams, with mean estimated occupancy probabilities ranging from 0.75 (95% CI 0.41-0.97) for *P. ruber* to 0.92 (95% CI 0.79-0.99) for *E. cirrigera* (Figure 3.2; See Appendix C.02 for mean estimated occupancy probabilities). Occupancy was lower at MTR/VF streams, with mean estimated occupancy probability ranging from 0.22 (95% CI 0.06-0.48) for *G. porphyriticus* to 0.60 (95% CI 0.33-0.86) for *D. fuscus* (Figure 3.2), although most occupancy estimates had large credible intervals. Despite these large credible intervals, we found that the species-specific $\alpha 1_i$ parameter estimates were negative and did not overlap with zero, which collectively indicate that all species were less likely to occupy MTR/VF streams. For our model, 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 & Rubin 1992).

Site-specific model-estimated number of species per stream for control sites ranged from 2.72 (95% CI 2.0-4.0) at Little Millseat B to 5.00 (95% CI 4.999-5.001) for Boardinghouse, Coles Fork, Falling Rock A, A Field, Little Millseat A, and Miller

(Figure 3.3). Site-specific estimated species richness for MTR/VF sites ranged from 0.14 (-0.001-1.0)) at Big Hollow to 5.00 (4.999-5.001)) at Bee Near (Figure 3.4). When all the salamander species were considered together, as a community the mean occupancy in MTR/VF streams was 0.35 (95% CI 0.02-0.94) and mean occupancy in control streams was 0.88 (95% CI 0.56-0.97) suggesting that salamanders have a higher probability of occupancy in streams that have not been affected by MTR/VF. The occupancy covariate (μ_{a1} MTR/VF) contained only negative values in the 95% credible interval -2.63 (95% CI -4.37-(-0.78)) and the standard deviation in the response to the covariate across species (σ_{a1} MTR/VF) contained only positive values 1.30 (95% CI 0.08-3.71), indicating certainty in the mean community response (Table 3.3). 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) contained positive and negative values in the 95% credible intervals, indicating uncertainty in the mean community responses to these covariates (Table 3.3). Model generated estimates of mean species richness indicated that control streams had greater mean species richness than MTR/VF sites (mean difference of 2.32 [95% CI 2.73-1.97)] between control and MTR/VF; Figure 3.5).

Table 3.1. Mean, 95% credible intervals (95% CI), and differences in environmental attributes at Mountain-top removal/valley fill and control (i.e., forest) intermittent streams located in the interior rugged section of the Cumberland Plateau in Breathitt and Knott Counties, Kentucky USA.

	MTR/VF		Control			
Variable	Mean	95% CI	Mean	95% CI	Difference	95% CI
Temperature (°C)	13.44	12.66-14.22	12.48	11.87-13.10	0.95	-0.03-(-1.95)
Forest cover (%)	0.25	0.12-0.38	0.997	0.993-0.999	-0.75	-0.88-(-0.62)
Specific conductivity (μS/cm)	1477.0	1103.0-1855.0	50.85	38.91-62.67	1427.0	1052.0-1804.0
Average stream width (cm)	122.6	88.33-156.7	130.6	102.3-159.1	-8.06	-52.47-35.54
Average stream depth (cm)	7.45	5.97-8.93	6.76	5.17-8.34	0.70	-1.47-2.85
Cover objects (#)	24.79	13.92-35.49	48.24	35.94-60.23	-23.45	-39.49-(-7.25)
Total organic carbon (mg/l)	7.97	2.63-13.47	2.76	1.86-3.66	5.204	-0.21-10.77
pH (H+)	6.08	5.35-6.82	5.71	5.34-6.09	0.3677	-0.45-1.18
SO_4 (mg/l)	506.7	260.1-758.2	7.22	3.47-10.99	499.5	252.9-751.3
Ca (mg/l)	23.72	21.79-25.65	1.28	1.10-1.45	22.44	20.51-24.38
Mg (mg/l)	10.14	9.75-10.54	1.62	1.40-1.83	8.526	8.08-8.97
K (mg/l)	8.15	6.04-10.26	2.11	1.08-3.13	6.043	3.72-8.40
Na (mg/l)	8.46	6.34-10.61	2.55	0.87-4.28	5.917	3.20-8.63

Table 3.2. Model estimated detection parameters and 95% credible intervals for each species observed at sites of mountaintop removal and natural second growth forest streams (Control) located in the interior rugged section of the Cumberland Plateau, Kentucky.

Species	Julian Date (CI)	Julian Date^2 (CI)	Water Temperature (CI)	Date of Last Precipitation (CI)
Desmognathus fuscus	-0.02 (-0.42-0.40)	0.31 (-0.10-0.90)	0.14 (-0.29-0.56)	-0.03 (-0.42-0.39)
Desmognathus monticola	0.11 (-0.29-0.62)	-0.06 (-0.52-0.30)	0.08 (-0.38-0.47)	0.13 (-0.27-0.69)
Pseudotriton ruber	-0.05 (-0.55-0.44)	0.11 (-0.32-0.59)	0.15 (-0.34-0.67)	-0.15 (-0.72-0.28)
Gyrinophilus porphyriticus	-0.02 (-0.49-0.47)	-0.04 (-0.54-0.36)	0.08 (-0.48-0.55)	-0.17 (-0.66-0.22)
Eurycea cirrigera	-0.26 (-0.82-0.16)	0.10 (-0.28-0.52)	0.23 (-0.15-0.71)	-0.13 (-0.58-0.25)

*Credible intervals are given in parenthesis (CI)

Table 3.3. Summary of hyper-parameters for occupancy and detection covariates for salamanders observed at sites of mountaintop removal and natural second growth forest streams (controls) located in the interior rugged section of the Cumberland Plateau, Kentucky. The symbol μ indicates mean community response, while σ indicates the standard deviation in the response to the covariate across species.

Community level hyper-parameter		Mean	Standard Deviation	95% Credible Interval	
$\mu_{\alpha 1}$	MTR/VF	-2.63	0.91	-4.37	-0.78
$\sigma_{\alpha 1}$	MTR/VF	1.30	0.91	0.08	3.71
$\mu_{\beta 1}$	Julian date	-0.05	0.25	-0.53	0.43
$\sigma_{\beta 1}$	Julian date	0.34	0.34	0.01	1.19
$\mu_{\beta 2}$	Julian date squared	0.09	0.25	-0.38	0.57
$\sigma_{\beta 2}$	Julian date squared	0.35	0.34	0.01	1.22
$\mu_{\beta 3}$	Water temperature	0.14	0.23	-0.30	0.56
$\sigma_{\beta 3}$	Water temperature	0.26	0.29	0.01	0.98
$\mu_{\beta 4}$	Date of last precipitation	-0.07	0.24	-0.54	0.38
$\sigma_{\beta 4}$	Date of last precipitation	0.32	0.33	0.01	1.16

Figure 3.1. Mean detection probabilities (±1 SD) of stream salamanders observed at sites of mountaintop removal (MTR/VF) and natural second growth forest streams (Control) located in the interior rugged section of the Cumberland Plateau, Kentucky.

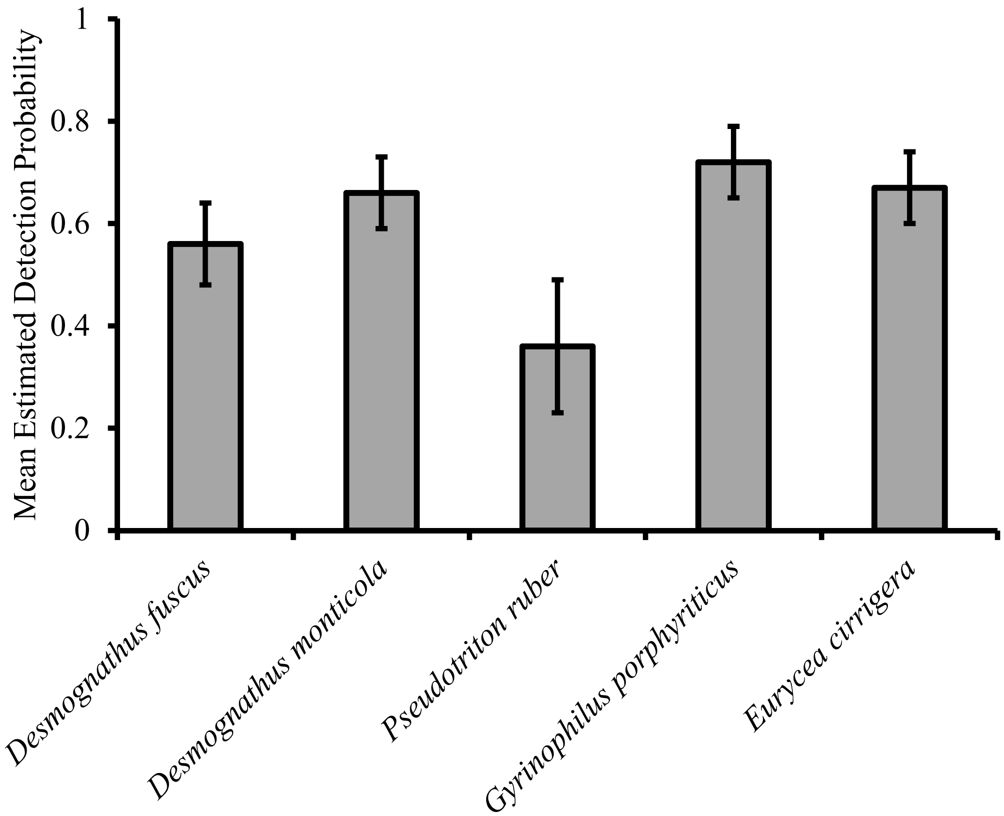


Figure 3.2. Mean estimated occupancy probabilities (±1 SD) of stream salamanders observed at sites of mountaintop removal (MTR/VF) and natural second growth forest streams (Control) located in the interior rugged section of the Cumberland Plateau, Kentucky.

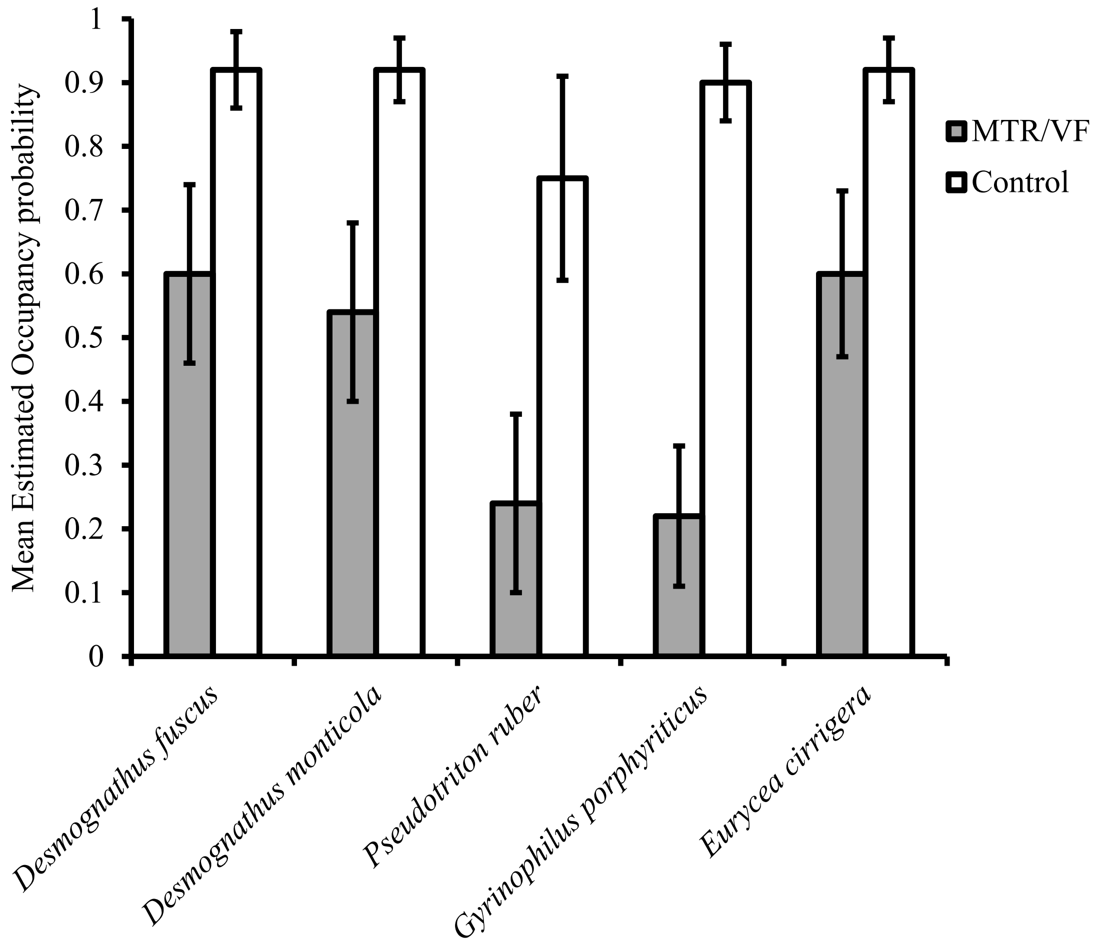


Figure 3.3. Site-specific estimated species richness (±1 SD) for control mature forest stream sites located in Breathitt and Knott counties, Kentucky on the Cumberland Plateau of Appalachia.

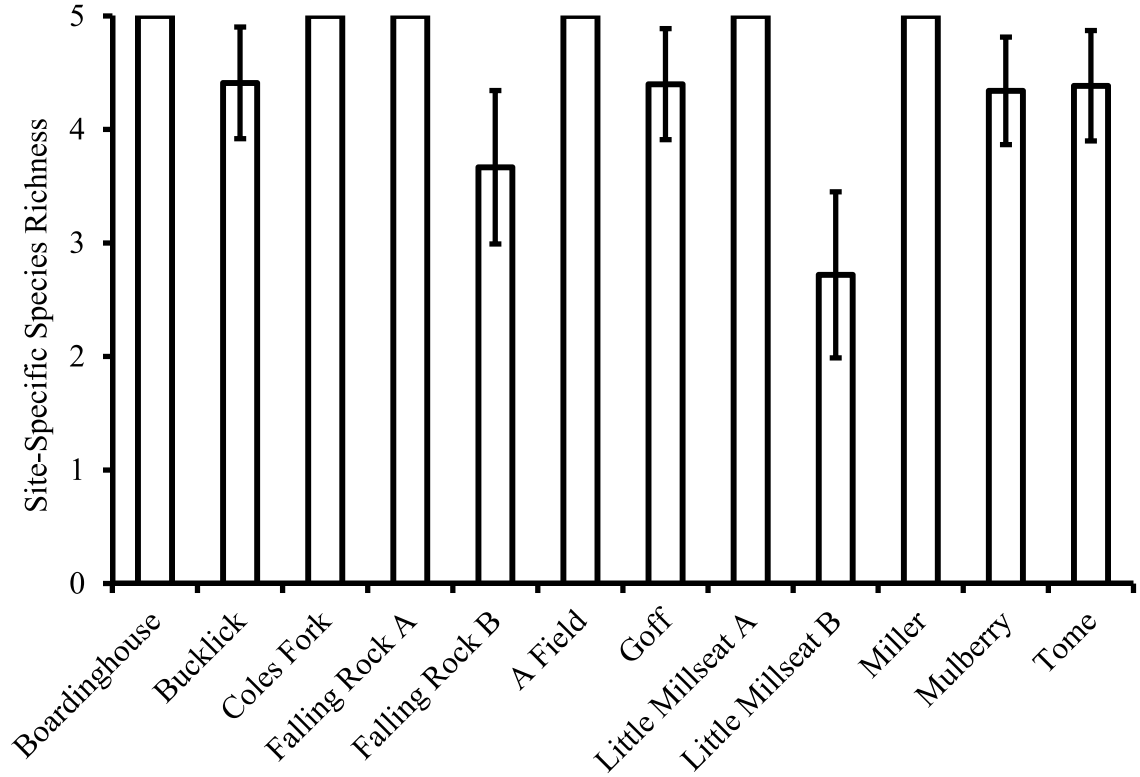


Figure 3.4. Site-specific estimated species richness (±1 SD) for MTR/VF stream sites located in Breathitt and Knott counties, Kentucky on the Cumberland Plateau of Appalachia.

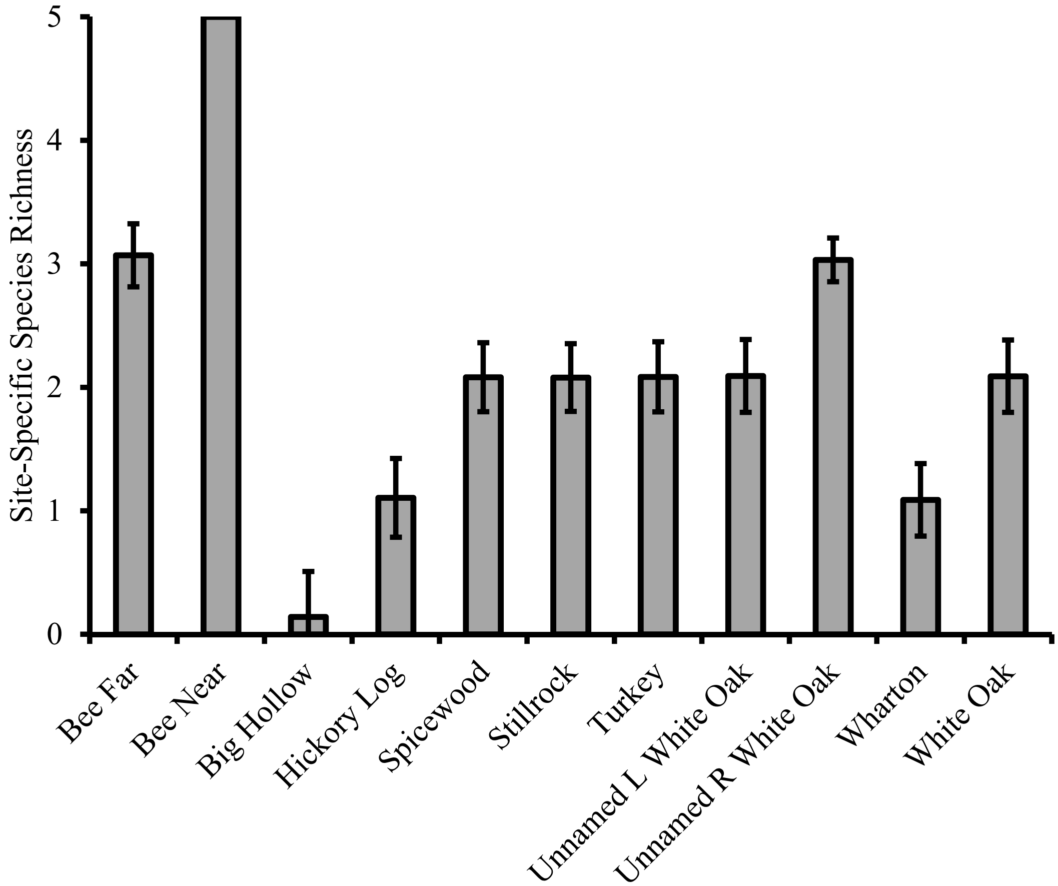
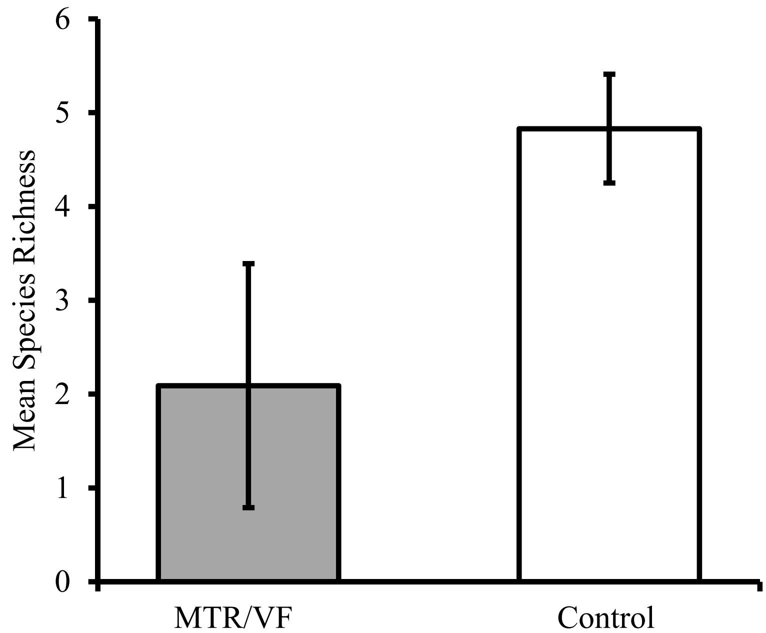


Figure 3.5. Estimated average species richness with standard deviation (±1 SD) for stream salamanders observed at sites of mountaintop removal (MTR/VF) and natural second growth forest streams (Control) located in the interior rugged section of the Cumberland Plateau, Kentucky.



CHAPTER 4: DISCUSSION

Our findings supported our hypothesis as we found that MTR/VF streams had reduced species salamander species richness and altered environmental attributes compared to control streams. Recent research in West Virginia found that stream salamander abundance was lower in MTR/VF streams, yet species richness did not differ between MTR/VF streams and control streams (Wood & Williams 2013). Based on our analysis, mean occupancy rates across five stream salamander species were reduced in MTR/VF compared to control sites. Because of this consistent response, we also observed decreased salamander species richness at MTR/VF streams in relation to control streams. These findings indicate that MTR/VF streams may not provide suitable habitat for stream salamander species in Central Appalachia and declines in salamander species richness due to MTR/VF are similar to those seen in invertebrate, mussel, and fish communities (Ferreri et al. 2004; Warren & Haag 2005; Pond 2010; Pond 2012).

Our estimates of mean salamander species' occupancy had relatively narrow credible intervals. Salamander species' occupancy at MTR/VF sites was lower for all species investigated, yet credible intervals were much broader and thus our occupancy estimates were less certain. Broad credible intervals for salamander species' occupancy estimates at MTR/VF streams likely resulted from differences in species' abundances between control streams and MTR/VF streams (Royle & Nichols 2003). Indeed, our area-constrained surveys at MTR/VF sites rarely resulted in the detection of ≥ 4 individual salamanders and a large proportion of sites where a species was first detected had no detections the next survey. We regularly detected ≥ 15 individuals per survey at control sites, and usually detected the same species nearly every survey. Royle and Nichols (2003) suggest that local abundance might be a major source of heterogeneity in detection probabilities, especially when local population sizes are small. Despite these broad credible intervals, we still found a statistically significant parameter estimate for MTR/VF on mean salamander occupancy (i.e., nonspecific specific) and species richness. However, we would expect increased precision as the number of sample locations and the number of repeated visits to those locations increase, if resources for further sampling were available (Dorazio & Royle 2005). Also, incorporating additional explanatory covariates for detection probability and occupancy in the models may reduce heterogeneity among sites and improve the precision of the estimates.

Reduced salamander occupancy rates and species richness at MTR/VF sites may be due to a complex set of interacting factors operating in both 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; Price et al. 2012). Indeed, the MTR/VF streams we studied had, on average, 75% less forest cover than control streams; land-cover within our catchments was dominated by non-native grasses and shrubs. Wood and Williams (2013) found 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.

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 species occupancy and species richness. Reclaimed mined sites have soils containing un-weathered, rock material, which is heavily compacted to reduce erosion, 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 salamanders (Barrett et al. 2010) and may influence survival and occupancy rates within MTR/VF streams. MTR/VF streams also have increased streamflow as a result of increased impervious surfaces (Ferrari et al. 2009), decreased infiltration (Chong & Cowsert 1997), and decreasing canopy interception and evapotranspiration (Messinger 2003). During the dormant season, streamflow is expected to be higher due to inhibited evapotranspiration compared to streamflow during the growing season (Zegre et al. 2013). Thus, due to lack of forested area at MTR/VF sites could lack seasonal variability in streamflow as seen in natural forested catchments. Lack of periods of high and low streamflow could influence occupancy rates within MTR/VF streams, particularly with higher flow rates.

Although we found little difference in stream width, depth, and forest cover within streams at reference and MTR/VF sites, stream bank erosion, and sedimentation can be excessive on MTR/VF streams (Fox 2009). Wood and Williams (2013) suggest that elevated siltation contributed to lower abundances of stream salamanders in West Virginia MTR/VF sites and Redmond (1980) found Black Mountain Dusky Salamanders (*D. welteri*) were excluded from highly silted streams due to coal mining. These results contrast with the findings of Keitzer and Goforth (2012) who found that the blue-ridge two-lined salamander (*E. wilderae*) abundance was not strongly affected by sedimentation. Thus, sediment alone may or may not be responsible for collective reduced occupancy and species richness seen at our MTR/VF sites.

We found MTR/VF streams had elevated levels of specific conductivity, total organic carbon, and dissolved ion concentrations. A previous study conducted at the Laurel Fork mine also found elevated specific conductivity 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 & Williams 2013). Amphibians are constantly transporting ions across their permeable skin to maintain water and ion balance (Ultsch et al. 1999), thus high specific conductivity may alter osmoregulation in amphibians due to their permeable skin where ion transportation constantly occurs and can increase corticosterone levels in amphibians (Chambers 2011), perhaps resulting in population declines and species extirpations. Further studies have demonstrated that elevated specific conductivity exposure negatively affects amphibian behavior (Chambers 2011), growth and development (Snodgrass et al. 2008), and survival (Sanzo & Hecnar 2006). Indeed, Miller et al. (2007) found that larval *E. cirrigera* abundance was negatively related to specific conductivity 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 conductivity levels (i.e., >100 µS/cm). Finally, decreases in macroinvertebrate species richness due to water chemistry are well documented in streams impacted by MTR/VF (Pond 2010; Pond 2012); stream invertebrates are an important prey item for salamanders (Petranka 1998; Davic & Welsh 2004). Thus, a reduction in prey items may also lead to reductions in salamander occupancy and species richness. The higher levels of TOC we observed at MTR/VF sites may be reflected by lack of macroinvertebrates, which are important for detritus breakdown and nutrient cycling (Merritt et al. 1984).

Despite the overall decreased occupancy and species richness of salamanders on MTR/VF streams, *D. fuscus*, *D. monticola* and *E. cirrigera* exhibited higher mean occupancy rates on MTR/VF streams than *P. ruber* and *G. porphyriticus*. Differences in estimated occupancy between species may reflect the species' life histories. In particular, both *G. porphyriticus* and *P. ruber* exhibit larval periods of ≥ 2 years (Bruce 1980; Petranka 1984; Bruce 1978); while species with higher occupancy rates (i.e. *D. fuscus, D. monticola, E. cirrigera*) often have shorter larval periods (Organ 1961). If larva exhibit elevated corticosterone levels and reduced prey consumption, as seen in Paybins et al. (2000) due to elevated specific conductivity, those with longer larval periods may have more difficulty persisting at MTR/VF sites, leading to reduced probability of occupancy.

The disturbance caused by MTR/VF is drastically changing the Appalachian landscape, compromising the natural ecological and functional state of both terrestrial and aquatic environments (Lindenmayer et al 2011). 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 long-lasting terrestrial impacts in combination with the valley fill process influences stream ecosystems. 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. However, restoration efforts, such as the Forestry Reclamation Approach (FRA) that advocate reforesting MTR/VF land, could be beneficial for salamander communities 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). Despite the patterns we documented in this study, research documenting the proximate mechanisms driving reduced salamander occupancy and species richness is needed.

APPENDIX A: MATRICES FOR ANALYISIS

A.01. Species encounter matrices. See Table 2.1 for site coordinates.

A.01.1. *Desmognathus fuscus* encounter matrix.

Site	MTR/VF / Control	Survey1	Survey2	Survey3	Survey4
Bee Far	MTR/VF	1	0	0	0
Bee Near	MTR/VF	1	1	0	1
Big Hollow	MTR/VF	0	0	0	0
Boardinghouse	Control	1	0	0	0
Bucklick Hollow	Control	1	1	1	1
Coles A	Control	0	0	1	1
Falling Rock A	Control	1	1	1	1
Falling Rock B	Control	0	0	0	0
A Field	Control	1	0	1	1
Goff Hollow	Control	1	1	1	1
Hickory Log	MTR/VF	0	0	0	0
Little Millseat A	Control	1	1	1	1
Little Millseat B	Control	1	1	0	1
Miller Hollow	Control	1	0	0	1
Mulberry	Control	1	1	0	1
Spicewood	MTR/VF	0	0	1	0
Stillrock	MTR/VF	0	0	1	0
Tome	Control	1	0	1	0
Turkey	MTR/VF	0	0	0	0
Unnamed L White Oak	MTR/VF	1	0	0	1
Unnamed R White Oak	MTR/VF	0	1	1	1
Wharton	MTR/VF	0	0	0	0
White Oak	MTR/VF	0	0	1	1

A.01.2. *Desmognathus monticola* encounter matrix.

Site	MTR/VF / Control	Survey1	Survey2	Survey3	Survey4
Bee Far	MTR/VF	0	0	1	1
Bee Near	MTR/VF	0	1	1	1
Big Hollow	MTR/VF	0	0	0	0
Boardinghouse	Control	1	1	1	1
Bucklick Hollow	Control	1	1	1	1
Coles A	Control	1	1	1	0
Falling Rock A	Control	1	0	0	0
Falling Rock B	Control	1	1	1	1
A Field	Control	1	1	1	1
Goff Hollow	Control	0	1	1	1
Hickory Log	MTR/VF	0	0	0	0
Little Millseat A	Control	0	1	1	1
Little Millseat B	Control	0	0	0	0
Miller Hollow	Control	1	1	1	1
Mulberry	Control	0	1	1	0
Spicewood	MTR/VF	0	0	0	0
Stillrock	MTR/VF	0	0	0	1
Tome	Control	1	1	1	1
Turkey	MTR/VF	0	0	0	1
Unnamed L White Oak	MTR/VF	1	0	0	0
Unnamed R White Oak	MTR/VF	0	0	0	0
Wharton	MTR/VF	0	1	0	0
White Oak	MTR/VF	0	0	0	0

A.01.3. *Pseudotriton ruber* encounter matrix.

Site	MTR/VF / Control	Survey1	Survey2	Survey3	Survey4
Bee Far	MTR/VF	0	0	0	0
Bee Near	MTR/VF	0	1	0	0
Big Hollow	MTR/VF	0	0	0	0
Boardinghouse	Control	1	0	1	1
Bucklick Hollow	Control	0	0	0	0
Coles A	Control	0	0	1	0
Falling Rock A	Control	0	0	1	1
Falling Rock B	Control	0	0	0	0
A Field	Control	1	1	0	1
Goff Hollow	Control	0	0	0	0
Hickory Log	MTR/VF	0	0	0	0
Little Millseat A	Control	0	1	0	0
Little Millseat B	Control	0	0	0	0
Miller Hollow	Control	1	0	0	0
Mulberry	Control	0	0	0	0
Spicewood	MTR/VF	0	0	0	0
Stillrock	MTR/VF	0	0	0	0
Tome	Control	0	0	0	0
Turkey	MTR/VF	0	0	0	0
Unnamed L White Oak	MTR/VF	0	0	0	0
Unnamed R White Oak	MTR/VF	0	1	1	0
Wharton	MTR/VF	0	0	0	0
White Oak	MTR/VF	0	0	0	0

A.01.4. *Gyrinophilus porphyriticus* encounter matrix.

Site	MTR/VF / Control	Survey1	Survey2	Survey3	Survey4
Bee Far	MTR/VF	0	0	0	0
Bee Near	MTR/VF	0	0	1	0
Big Hollow	MTR/VF	0	0	0	0
Boardinghouse	Control	1	1	1	1
Bucklick Hollow	Control	1	1	1	0
Coles A	Control	1	1	1	0
Falling Rock A	Control	1	0	1	1
Falling Rock B	Control	1	1	1	1
A Field	Control	1	1	1	1
Goff Hollow	Control	1	0	1	0
Hickory Log	MTR/VF	0	0	0	0
Little Millseat A	Control	0	1	0	1
Little Millseat B	Control	1	1	0	0
Miller Hollow	Control	0	1	1	1
Mulberry	Control	0	1	1	1
Spicewood	MTR/VF	0	0	0	0
Stillrock	MTR/VF	0	0	0	0
Tome	Control	1	1	1	1
Turkey	MTR/VF	0	0	0	0
Unnamed L White Oak	MTR/VF	0	0	0	0
Unnamed R White Oak	MTR/VF	0	0	0	0
Wharton	MTR/VF	0	0	0	0
White Oak	MTR/VF	0	0	0	0

A.01.5. *Eurycea cirrigera* encounter matrix.

Site	MTR/VF / Control	Survey1	Survey2	Survey3	Survey4
Bee Far	MTR/VF	0	1	0	0
Bee Near	MTR/VF	1	1	0	0
Big Hollow	MTR/VF	0	0	0	0
Boardinghouse	Control	1	1	1	1
Bucklick Hollow	Control	1	1	1	1
Coles A	Control	1	0	1	0
Falling Rock A	Control	1	1	1	1
Falling Rock B	Control	1	1	1	1
A Field	Control	1	1	1	1
Goff Hollow	Control	1	1	1	1
Hickory Log	MTR/VF	0	1	1	1
Little Millseat A	Control	1	1	1	1
Little Millseat B	Control	0	0	0	0
Miller Hollow	Control	1	0	0	1
Mulberry	Control	1	1	1	1
Spicewood	MTR/VF	0	1	0	0
Stillrock	MTR/VF	0	0	0	0
Tome	Control	1	0	1	1
Turkey	MTR/VF	1	0	0	0
Unnamed L White Oak	MTR/VF	0	0	0	0
Unnamed R White Oak	MTR/VF	1	0	0	0
Wharton	MTR/VF	0	0	0	0
White Oak	MTR/VF	1	1	0	0

A.02. Sampling covariate matrices. See Table 2.1 for site coordinates.

A.02.1. Day of year sampling covariate matrix.

Site	MTR/VF / Control	DY1	DY2	DY3	DY4
Bee Far	MTR/VF	103	132	156	174
Bee Near	MTR/VF	103	132	156	174
Big Hollow	MTR/VF	108	128	156	175
Boardinghouse	Control	73	127	144	171
Bucklick Hollow	Control	123	135	155	172
Coles A	Control	123	135	172	172
Falling Rock A	Control	116	127	155	171
Falling Rock B	Control	116	127	155	171
A Field	Control	121	129	143	164
Goff Hollow	Control	101	127	142	171
Hickory Log	MTR/VF	108	128	156	175
Little Millseat A	Control	101	129	143	164
Little Millseat B	Control	103	129	143	164
Miller Hollow	Control	101	127	142	171
Mulberry	Control	102	127	142	171
Spicewood	MTR/VF	113	129	156	175
Stillrock	MTR/VF	108	128	144	176
Tome	Control	121	129	142	171
Turkey	MTR/VF	113	129	156	175
Unnamed L White Oak	MTR/VF	113	128	144	176
Unnamed R White Oak	MTR/VF	113	128	144	176
Wharton	MTR/VF	108	128	156	175
White Oak	MTR/VF	113	128	144	176

A.02.2. Water temperature sampling covariate matrix of ized values.

Site	MTR/VF / Control	Temp1	Temp2	Temp3	Temp4
Bee Far	MTR/VF	-1.23	0.24	-0.83	-1.03
Bee Near	MTR/VF	0.08	1.34	0.20	-0.02
Big Hollow	MTR/VF	1.01	0.29	-0.16	-1.67
Boardinghouse	Control	-3.02	-1.36	-0.58	0.18
Bucklick Hollow	Control	-0.14	1.41	0.95	0.04
Coles A	Control	-0.01	-0.40	0.56	-0.43
Falling Rock A	Control	-0.45	-1.12	0.78	-0.10
Falling Rock B	Control	-0.60	-0.86	0.10	-0.85
A Field	Control	-0.09	0.10	0.45	0.92
Goff Hollow	Control	-0.25	-1.11	0.05	-0.92
Hickory Log	MTR/VF	2.00	2.21	0.64	0.92
Little Millseat A	Control	1.17	0.39	0.63	1.86
Little Millseat B	Control	-0.79	-0.43	-0.35	0.59
Miller Hollow	Control	-0.66	-1.47	-0.65	-1.25
Mulberry	Control	-0.10	-0.24	1.40	0.72
Spicewood	MTR/VF	0.89	0.15	1.51	0.03
Stillrock	MTR/VF	0.26	-0.05	-1.91	0.51
Tome	Control	0.21	0.07	0.17	-0.19
Turkey	MTR/VF	0.34	-1.02	1.37	2.06
Unnamed L White Oak	MTR/VF	-0.17	-0.44	-1.76	0.77
Unnamed R White Oak	MTR/VF	-0.38	-0.08	-1.48	-1.73
Wharton	MTR/VF	1.24	2.00	0.42	0.31
White Oak	MTR/VF	0.69	0.37	-1.51	-0.72

A.02.3. Date of last precipitation sampling covariate matrix of standardized values.

Site	MTR/VF / Control	DOP1	DOP2	DOP3	DOP4
Bee Far	MTR/VF	-0.60	-0.30	1.66	-0.09
Bee Near	MTR/VF	-0.60	-0.30	1.66	-0.09
Big Hollow	MTR/VF	-1.19	-0.30	-1.66	-0.09
Boardinghouse	Control	0.00	-0.30	0.00	-0.09
Bucklick Hollow	Control	0.60	3.17	0.00	2.02
Coles A	Control	0.60	3.17	1.66	2.02
Falling Rock A	Control	-0.60	-0.30	0.00	-0.09
Falling Rock B	Control	-0.60	-0.30	0.00	-0.09
A Field	Control	0.00	-0.30	0.00	-2.20
Goff Hollow	Control	1.79	-0.30	0.00	-0.09
Hickory Log	MTR/VF	-1.19	-0.30	-1.66	-0.09
Little Millseat A	Control	1.79	-0.30	0.00	-2.20
Little Millseat B	Control	-1.19	-0.30	0.00	2.02
Miller Hollow	Control	1.79	-0.30	0.00	-0.09
Mulberry	Control	-1.19	-0.30	0.00	-0.09
Spicewood	MTR/VF	0.60	-0.30	1.66	-0.09
Stillrock	MTR/VF	-1.19	-0.30	0.00	-0.09
Tome	Control	0.00	-0.30	0.00	-0.09
Turkey	MTR/VF	0.60	-0.30	-1.66	-0.09
Unnamed L White Oak	MTR/VF	0.60	-0.30	0.00	-0.09
Unnamed R White Oak	MTR/VF	0.60	-0.30	0.00	-0.09
Wharton	MTR/VF	-1.19	-0.30	-1.66	-0.09
White Oak	MTR/VF	0.60	-0.30	0.00	-0.09

APPENDIX B: PROGRAM R CODES

B.01. R code used for occupancy and site-specific species richness analysis.

```
## Load packages
library(R2OpenBUGS)

### Import and prepare data for use in function
# species data: rows = sites, columns = sampling occasions
desfus<-read.csv('desfus.csv')
desfus<-as.matrix(desfus[,-1])

desmon<-read.csv('desmon.csv')
desmon<-as.matrix(desmon[,-1])

psurub<-read.csv('psurub.csv')
psurub<-as.matrix(psurub[,-1])

gyrpor<-read.csv('gyrpor.csv')
gyrpor<-as.matrix(gyrpor[,-1])

eurcir<-read.csv('eurcir.csv')
eurcir<-as.matrix(eurcir[,-1])

## build array where structure is [site,sample,species]
```

```
X<-array(data=c(desfus,desmon,psurub,gyrpor,eurcir),dim=c(23,4,5))


## site covariates

SiteCovMined<-read.csv('SiteCovariates.csv') # These are standardized in the
accompanying .csv file, mean is centered around 0

# survey covariates (plus standardization)


jd<-read.csv('SamplingCovDOY.csv')[,1:5]

jd<-as.matrix(jd[,-1])

jd<-(jd-mean(jd))/sd(as.vector(jd))

jd2<-jd^2

#jd<-read.csv('SamplingCovJDate.csv')

#jd<-as.matrix(jd[,-1])

#jd<-(jd-mean(jd))/sd(as.vector(jd))

#jd2<-jd^2


temp<-read.csv('SamplingCovWaterTemp.csv')

temp<-as.matrix(temp[,-1])

temp<-(temp-mean(temp))/sd(as.vector(temp))


DoP<-read.csv('SamplingCovDOP.csv')

DoP<-as.matrix(DoP[,-1])

DoP<-(DoP-mean(DoP))/sd(as.vector(DoP))
```

No augmentation - BJH did not run this one

```
BMuncyOccupancy<-
function(DH=X,Mined=SiteCovMined$Mined,jd=jd,jd2=jd2,temp=temp,DoP=DoP,nc=3
,ni=300000,nb=30000,nt=3){

 n<-dim(DH)[3] # number of species

 J<-dim(DH)[1] # number of sites = rows

 K<-dim(DH)[2] # number of occassions = columns

 data<-list(Y=DH,n=n,J=J,K=K,Mined=Mined,jd=jd,jd2=jd2,temp=temp,DoP=DoP)

 modelFilename = "MultiSpeciesOccModel.txt"

 cat("

 model{

  psi.mean~dunif(0,1)

  a<-log(psi.mean)-log(1-psi.mean)

  p.mean~dunif(0,1)

  b<-log(p.mean)-log(1-p.mean)

  sigma.u~dunif(0,5)

  sigma.v~dunif(0,5)

  tau.u<-pow(sigma.u,-2)

  tau.v<-pow(sigma.v,-2)

  rho~dunif(-1,1)

  var.v<-tau.v/(1-pow(rho,2))

  mu.alpha1~dunif(-10,10)

  mu.beta1~dunif(-10,10)
```

```
mu.beta2~dunif(-10,10)
mu.beta3~dunif(-10,10)
mu.beta4~dunif(-10,10)
sigma.alpha1~dunif(0,5)
sigma.beta1~dunif(0,5)
sigma.beta2~dunif(0,5)
sigma.beta3~dunif(0,5)
sigma.beta4~dunif(0,5)
tau.alpha1<-1/pow(sigma.alpha1,2)
tau.beta1<-1/pow(sigma.beta1,2)
tau.beta2<-1/pow(sigma.beta2,2)
tau.beta3<-1/pow(sigma.beta3,2)
tau.beta4<-1/pow(sigma.beta4,2)
for(i in 1:n){
 u[i]~dnorm(a,tau.u)I(-5,5)
 mu.v[i]<-b+(rho*sigma.v/sigma.u)*(u[i]-a)
 v[i]~dnorm(mu.v[i],var.v)I(-10,5)
 alpha1[i]~dnorm(mu.alpha1, tau.alpha1)I(-5,5)
 beta1[i]~dnorm(mu.beta1, tau.beta1)I(-5,5)
 beta2[i]~dnorm(mu.beta2, tau.beta2)I(-5,5)
 beta3[i]~dnorm(mu.beta3, tau.beta3)I(-5,5)
 beta4[i]~dnorm(mu.beta4, tau.beta4)I(-5,5)
 #Estimate the occupancy probability (latent Z matrix) for each species at each site
 for(j in 1:J){
```

```
      logit(psi[j,i])<-u[i]+alpha1[i]*Mined[j]

      Z[j,i]~dbern(psi[j,i])

      #Estimate the species specific detection probability for every rep at each point where the species occurs (Z=1)

      for(k in 1:K){

       logit(p[j,k,i])<-v[i]+beta1[i]*jd[j,k]+ beta2[i]*jd2[j,k]+beta3[i]*temp[j,k]+beta4[i]*DoP[j,k]

       mu.p[j,k,i]<-p[j,k,i]*Z[j,i]

       Y[j,k,i]~dbern(mu.p[j,k,i])

       #Create simulated dataset to calculate the Bayesian p-value

       Ynew[j,k,i]~dbern(mu.p[j,k,i])

       d[j,k,i]<-abs(Y[j,k,i]-mu.p[j,k,i])

       dnew[j,k,i]<-abs(Ynew[j,k,i]-mu.p[j,k,i])

       d2[j,k,i]<-pow(d[j,k,i],2)

       dnew2[j,k,i]<-pow(dnew[j,k,i],2)

      }

      dsum[j,i]<-sum(d2[j,1:K,i])

      dnewsum[j,i]<-sum(dnew2[j,1:K,i])

     }

   }

   #Calculate the discrepancy measure, which is then defined as the mean(p.fit > p.fitnew)

   P.fit<-sum(dsum[1:J,1:n])

   P.fitnew<-sum(dnewsum[1:J,1:n])

   BPvalue<-step(P.fitnew-P.fit)
```

```
    # derived parameters
    for(j in 1:J){
      SpR[j]<-sum(Z[j,])
    }
  }
  ", fill=TRUE, file=modelFilename)
  inits<-function(){

list(psi.mean=runif(1),p.mean=runif(1),sigma.u=runif(1,0,5),sigma.v=runif(1,0,5),rho=ru
nif(1,-1,1),u=rnorm(n),v=rnorm(n),Z=matrix(1,ncol=n,nrow=J),
      mu.alpha1=rnorm(1),sigma.alpha1=runif(1,0,5),

mu.beta1=rnorm(1),mu.beta2=rnorm(1),mu.beta3=rnorm(1),mu.beta4=rnorm(1),sigma.b
eta1=runif(1,0,5),sigma.beta2=runif(1,0,5),sigma.beta3=runif(1,0,5),sigma.beta4=runif(1,
0,5))
  }
  params<-list('psi.mean','sigma.u','p.mean','sigma.v','SpR','mu.alpha1','sigma.alpha1',

'alpha1','mu.beta1','mu.beta2','mu.beta3','mu.beta4','sigma.beta1','sigma.beta2','sigma.beta
3','sigma.beta4','beta1','beta2','beta3','beta4','BPvalue','u','a','v')


  # library(R2WinBUGS)
  fit<-
bugs(data,inits,params,"MultiSpeciesOccModel.txt",n.chains=nc,n.iter=ni,n.burnin=nb,n.
thin=nt,debug=TRUE,DIC=TRUE, working.directory=getwd())
  print(fit,dig=3)
}
```

```
### Run function

BMuncyOccupancyFunction <-

   BMuncyOccupancy(DH=X,

                 Mined=SiteCovMined$Mined,

                 jd=jd,jd2=jd2,temp=temp,DoP=DoP,

                 nc=3,

                 ni=300000,

                 nb=30000,

                 nt=3)


                                   # Save to text file

sink('BMuncyOccupancyFunction.txt')

print(BMuncyOccupancyFunction,dig=3)

sink()


# Calculate Bayesian p-value (didn't have Kery book with me - there's probably a more efficient way to do it, but this works)

bpint<-rep(NA,100000)

for(i in 1:100000){

  bpint[i]<-if((BMuncyOccupancyFunction$sims.list$P.fit[i]-
BMuncyOccupancyFunction$sims.list$P.fitnew[i])>0) 1 else 0

}

BPvalue<-mean(bpint)
```

```
##### Compute posterior statisitics for occupancy probabilities #####
library(coda)

## Extract linear predictor values on logit scale
## 1) Forest
logit.psi.forest <- BMuncyOccupancyFunction$sims.matrix[,paste0("u[",1:5,"]")]
colnames(logit.psi.forest) <- c("Desfus","Desmon","Pserub","Gyrpor","Eurcir")

## 2) Mined
logit.psi.mined <- logit.psi.forest +
   BMuncyOccupancyFunction$sims.matrix[,paste0("alpha1[",1:5,"]")]
colnames(logit.psi.mined) <- c("Desfus","Desmon","Pserub","Gyrpor","Eurcir")

## Transform to probability scale
expit <- function(x) 1/(1 + exp(-x))

psi.forest <- expit(logit.psi.forest)
psi.mined <- expit(logit.psi.mined)

## Compute summary statistics for occupancy probabilities
## 1) Forest sites
(psi.forest.summ <- summary(as.mcmc(psi.forest)))
```

```
## 2) Mined sites
(psi.mined.summ <- summary(as.mcmc(psi.mined)))


## Extract posterior means and 95\% CI
## 1) Forest sites
round(cbind(psi.forest.summ[[1]][,"Mean"],
        psi.forest.summ[[2]][,c("2.5%","97.5%")]),2)


## 2) Mined sites
round(cbind(psi.mined.summ[[1]][,"Mean"],
        psi.mined.summ[[2]][,c("2.5%","97.5%")]),2)
```

B.02. R code for estimated average species richness across all sites.

```
###Load package plotrix

## Calculate differences in mean species richness among habitats

# set up empty vectors for the results

S.mine<-rep(NA,length(BMuncyOccupancyFunction$sims.list$SpR[,1]))

S.forest<-rep(NA,length(BMuncyOccupancyFunction$sims.list$SpR[,1]))

Sd.forest<-rep(NA, length(BMuncyOccupancyFunction$sims.list$SpR[,1]))

Sd.mine<-rep(NA, length(BMuncyOccupancyFunction$sims.list$SpR[,1]))

Se.forest<-rep(NA, length(BMuncyOccupancyFunction$sims.list$SpR[,1]))

Se.mine<-rep(NA, length(BMuncyOccupancyFunction$sims.list$SpR[,1]))



diff.mf<-rep(0,length(BMuncyOccupancyFunction$sims.list$SpR[,1]))

greater.mf<-rep(NA,length(BMuncyOccupancyFunction$sims.list$SpR[,1]))

# loop over i iterations#

for(i in 1:length(BMuncyOccupancyFunction$sims.list$SpR[,1])){

  # calculate mean species richness in each habitat type
```

```
  S.mine[i]<-
mean(c(BMuncyOccupancyFunction$sims.list$SpR[i,1],BMuncyOccupancyFunction$si
ms.list$SpR[i,2],BMuncyOccupancyFunction$sims.list$SpR[i,3],
BMuncyOccupancyFunction$sims.list$SpR[i,11],BMuncyOccupancyFunction$sims.list$
SpR[i,16],BMuncyOccupancyFunction$sims.list$SpR[i,17],
BMuncyOccupancyFunction$sims.list$SpR[i,19],BMuncyOccupancyFunction$sims.list$
SpR[i,20],BMuncyOccupancyFunction$sims.list$SpR[i,21],
BMuncyOccupancyFunction$sims.list$SpR[i,22],BMuncyOccupancyFunction$sims.list$
SpR[i,23]),na.rm=T)
  S.forest[i]<-
mean(c(BMuncyOccupancyFunction$sims.list$SpR[i,4],BMuncyOccupancyFunction$si
ms.list$SpR[i,5],BMuncyOccupancyFunction$sims.list$SpR[i,6],
BMuncyOccupancyFunction$sims.list$SpR[i,7],BMuncyOccupancyFunction$sims.list$S
pR[i,8],BMuncyOccupancyFunction$sims.list$SpR[i,9],
BMuncyOccupancyFunction$sims.list$SpR[i,10],BMuncyOccupancyFunction$sims.list$
SpR[i,12],BMuncyOccupancyFunction$sims.list$SpR[i,13],
BMuncyOccupancyFunction$sims.list$SpR[i,14],BMuncyOccupancyFunction$sims.list$
SpR[i,15],BMuncyOccupancyFunction$sims.list$SpR[i,18]),na.rm=T)
  # calculate pair-wise differences in mean species richness
diff.mf[i]<-S.mine[i]-S.forest[i]
```

```
  # indicator for if each difference is positive or negative
greater.mf[i]<-if(diff.mf[i]>0) 1 else 0
}
## produce posterior results of interest
# mean and 95% CI for differences of differences in S - these are better than comparing posterior credible intervals directly
# mine vs. forest
mean(diff.mf)
quantile(diff.mf,probs=c(0.025,0.975))
# posterior probability that one habitat has greater S than another - no credible intervals for these
# mine vs. forest
mean(greater.mf)
S.mine[i] #average spp richness at mine sites
S.forest[i] #average spp richness at forest sites
```

```
Sd.forest[i]<-
sd(c(BMuncyOccupancyFunction$sims.list$SpR[i,4],BMuncyOccupancyFunction$sims.l
ist$SpR[i,5],BMuncyOccupancyFunction$sims.list$SpR[i,6],
BMuncyOccupancyFunction$sims.list$SpR[i,7],BMuncyOccupancyFunction$sims.list$S
pR[i,8],BMuncyOccupancyFunction$sims.list$SpR[i,9],
BMuncyOccupancyFunction$sims.list$SpR[i,10],BMuncyOccupancyFunction$sims.list$
SpR[i,12],BMuncyOccupancyFunction$sims.list$SpR[i,13],
BMuncyOccupancyFunction$sims.list$SpR[i,14],BMuncyOccupancyFunction$sims.list$
SpR[i,15],BMuncyOccupancyFunction$sims.list$SpR[i,18]),na.rm=T)
Sd.mine[i]<-
sd(c(BMuncyOccupancyFunction$sims.list$SpR[i,1],BMuncyOccupancyFunction$sims.l
ist$SpR[i,2],BMuncyOccupancyFunction$sims.list$SpR[i,3],
BMuncyOccupancyFunction$sims.list$SpR[i,11],BMuncyOccupancyFunction$sims.list$
SpR[i,16],BMuncyOccupancyFunction$sims.list$SpR[i,17],
BMuncyOccupancyFunction$sims.list$SpR[i,19],BMuncyOccupancyFunction$sims.list$
SpR[i,20],BMuncyOccupancyFunction$sims.list$SpR[i,21],
BMuncyOccupancyFunction$sims.list$SpR[i,22],BMuncyOccupancyFunction$sims.list$
SpR[i,23]),na.rm=T)
Se.forest[i]<-
std.error(c(BMuncyOccupancyFunction$sims.list$SpR[i,4],BMuncyOccupancyFunction
```

```
$sims.list$SpR[i,5],BMuncyOccupancyFunction$sims.list$SpR[i,6],
BMuncyOccupancyFunction$sims.list$SpR[i,7],BMuncyOccupancyFunction$sims.list$S
pR[i,8],BMuncyOccupancyFunction$sims.list$SpR[i,9],
BMuncyOccupancyFunction$sims.list$SpR[i,10],BMuncyOccupancyFunction$sims.list$
SpR[i,12],BMuncyOccupancyFunction$sims.list$SpR[i,13],
BMuncyOccupancyFunction$sims.list$SpR[i,14],BMuncyOccupancyFunction$sims.list$
SpR[i,15],BMuncyOccupancyFunction$sims.list$SpR[i,18]),na.rm=T)
Se.mine[i]<-
std.error(c(BMuncyOccupancyFunction$sims.list$SpR[i,1],BMuncyOccupancyFunction
$sims.list$SpR[i,2],BMuncyOccupancyFunction$sims.list$SpR[i,3],
BMuncyOccupancyFunction$sims.list$SpR[i,11],BMuncyOccupancyFunction$sims.list$
SpR[i,16],BMuncyOccupancyFunction$sims.list$SpR[i,17],
BMuncyOccupancyFunction$sims.list$SpR[i,19],BMuncyOccupancyFunction$sims.list$
SpR[i,20],BMuncyOccupancyFunction$sims.list$SpR[i,21],
BMuncyOccupancyFunction$sims.list$SpR[i,22],BMuncyOccupancyFunction$sims.list$
SpR[i,23]),na.rm=T)
```

APPENDIX C: MEAN ESTIMATES

C.01. Mean detection probabilities (±1 SD and 95% credible intervals [CI]) of stream salamanders observed at sites of mountaintop removal (MTR/VF) and natural second growth forest streams (Control) located in the interior rugged section of the Cumberland Plateau, Kentucky.

Species	Mean Estimated Detection	SD	95% Credible Interval
Desmognathus fuscus	0.56	0.08	0.40-0.70
Desmognathus monticola	0.66	0.07	0.52-0.80
Pseudotriton ruber	0.36	0.13	0.15-0.62
Gyrinophilus porphyriticus	0.72	0.07	0.58-0.86
Eurycea cirrigera	0.67	0.07	0.53-0.80

C.02. Mean estimated occupancy probabilities (±1 SD and 95% credible intervals [CI]) of stream salamanders observed at sites of mountaintop removal (MTR/VF) and natural second growth forest streams (Control) located in the interior rugged section of the Cumberland Plateau, Kentucky.

	MTR/VF			Control		
Species	Mean Estimated Occupancy	SD	95% Credible Interval	Mean Estimated Occupancy	SD	95% Credible Interval
Desmognathus fuscus	0.60	0.14	0.33-0.86	0.92	0.06	0.78-0.99
Desmognathus monticola	0.54	0.14	0.28-0.80	0.92	0.05	0.78-0.99
Pseudotriton ruber	0.24	0.14	0.04-0.59	0.75	0.16	0.41-0.97
Gyrinophilus porphyriticus	0.22	0.11	0.06-0.47	0.90	0.06	0.74-0.98
Eurycea cirrigera	0.60	0.13	0.34-0.85	0.92	0.05	0.79-0.99

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