The Effects of Mountaintop Removal Mining and Valley Fills on Stream Salamander Occupancy and Community Composition

Brenee' L. Muncy[a], Steven J. Price[a], Simon J. Bonner[b] and Christopher D. Barton[a]

[a]Department of Forestry, University of Kentucky, Lexington, KY 40546-0073 USA

[b]Department of Statistics, University of Kentucky, Lexington, KY 40546-0082 USA

Corresponding Author (SJP) email: steven.price@uky.edu

Phone: 859-257-7610; Fax: 859-323-1031

Abstract

Mountaintop removal mining and valley filling (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 probabilities and overall community composition 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 0.41-0.96) to 0.90 (95% CI 0.77-0.98). Occupancy was lower at MTR/VF streams, with mean estimated occupancy probability ranging from 0.23 (95% CI 0.04-0.51) to 0.62 (95% CI 0.36-0.86). 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 (SO_4) 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 salamander occupancy and communities are reduced in streams impacted by MTR/VF mining practices.

Abbreviations

MTR/VF, mountaintop removal mining and valley filling; Ψ, occupancy; Θ, detection probability; u_i , species-specific mean probability of occurrence; v_i, species-specific mean probability of detection; U, uniform distribution

Keywords: amphibians; Appalachia; coal mining; Kentucky; species richness

Introduction

Mountaintop removal has become the dominant type of mining for the extraction of shallow coal seams in central 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 conductance, and elevated levels of total dissolved solids (i.e., sulfates (SO_4), 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 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 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 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

richness in MTR/VF streams is significantly reduced compared to reference locations (Pond, 2010, 2012), and decreases in freshwater mussel diversity are positively correlated with extent of surface mines 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 (2013) 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.

Methods

82 ***Study Sites –***

83 We investigated salamander occupancy probabilities and community composition at 23
84 first-order streams located in the interior rugged section of the Cumberland Plateau in Breathitt
85 and Knott Counties, Kentucky USA. This region has seen extensive changes in land-use over the
86 last 30 years; more than 194,000 ha of eastern Kentucky has been affected by surface mining (C.
87 Barton, personal communication). We sampled salamanders at 11 MTR/VF first-order streams
88 located on the reclaimed Laurel Fork surface mine (4144091.438 N 307635.435 E Zone 17) and
89 12 control first-order streams in approximately 80-yr-old ,second-growth forest on the
90 University of Kentucky's Robinson Forest, which shares a northeast border with the Laurel Fork
91 surface mine. Robinson Forest is a 5,983 ha teaching, research and extension experimental forest
92 composed of eight discontinuous properties. Our control streams were located with the main
93 block of Robinson Forest comprising approximately 4,200 ha. Land-cover with catchments of
94 control streams consisted of typical, mixed mesophytic forests of the region; dominant tree
95 species consisted of white oak (*Quercus alba*), tulip tree (*Liriodendron tulipifera*), Eastern
96 hemlock (*Tsuga canadensis*), and chestnut oak (*Quercus prinus*) (See Phillippi and Boebinger,
97 1986).

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

Data Collection Methods –

Area-constrained active searches were used to sample salamanders at each stream, in a single, 10-m sampling transect. We chose transects within streams 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. 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 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 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

128 counted the number of cover objects within our sampling transects. Specifically, we considered
129 rocks > 50 mm in diameter as well as logs and other woody debris cover objects of importance to
130 salamanders. Also, we recorded air temperature (C°), water temperature (C°), wind speed, degree
131 of cloudiness, and the date of last precipitation. Additionally, a 50 mL water sample was
132 collected prior to each sampling event and placed on ice. The samples were analyzed for
133 concentrations of Ca, Mg, SO_4^{2-}, potassium (K), sodium (Na), total organic carbon (TOC), pH
134 and specific conductance; sampling, preservation, and analytic protocols were performed in
135 accordance with standard methods (Greenberg et al., 1992).

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

143 ***Data Analysis –***

144 We used Bayesian t-tests with unequal variances (Kèry, 2010) to compare several
145 environmental attributes between control and MTR/VF streams. Attributes included: percent of
146 the stream catchment in forest cover, average stream wetted width and depth in our sampling
147 transects, number of cover objects within our sampling transects, water temperature, specific
148 conductance, TOC, pH, SO_4, Ca, Mg, K, and Na. All water quality data used in the analysis were
149 obtained during May 1-15, 2013 salamander sampling events. We used uninformative priors for
150 each model, which varied depending on the covariate being analyzed (i.e., percent forest cover

151 mean = Uniform distribution (U(0, 1)), standard deviation (SD) = U(0, 10); average stream width
152 mean = U(0, 250), SD U =(0,300); average stream depth mean = U(0, 25), SD = U(0,30); cover
153 objects mean = U(0,80), SD = U(0,100); water temperature mean = U(0, 25), SD U(0, 30);
154 specific conductance mean = U(0, 3000), SD U(0, 10000); TOC mean = U(0, 100), SD =
155 U(0,500); pH mean = U(0, 10), SD = U(0, 15); SO_4 mean = U(0, 1500), SD = U(0, 2000); Ca
156 mean = U(0, 50), SD = U(0, 75); Mg, K, and Na mean =U(0, 20), SD = U(0, 30)). We used the
157 R add-in library R2OpenBUGS (Sturtz et al., 2005), to organize our data into program R (2.14.0)
158 (R Development Core Team, 2010), and used Markov chain Monte Carlo methods as
159 implemented in OpenBUGS (Lunn et al., 2009) with three chains of 20,000 iterations, thinning
160 factor of 1 after 5,000 burn-in iterations to analyze each model. We evaluated the Markov chains
161 by examining the history plots and the Gelman-Rubin statistic for each parameter for evidence of
162 lack of convergence (Gelman and Rubin, 1992). The Gelman-Rubin statistic compares between-
163 and within-chain variability; values near 1 (and up to 1.1) indicate likely convergence (Gelman
164 and Hill, 2007).

165 We used a hierarchical Bayesian modeling approach to estimate species-specific and
166 community responses to MTR/VF mining. This multi-level approach provided estimates of site-
167 specific species richness in addition to separate estimates for species-specific occupancy and
168 detection probabilities; therefore community-level and species-level attributes are incorporated
169 into the same modeling framework (Dorazio and Royle, 2005; Zipkin et al., 2009). Specifically,
170 we used a model similar to that used by Zipkin et al. (2009) and Hunt et al. (2013), to estimate
171 species' occupancy and community responses to one site covariate (i.e., *MTR/VF*) and four
172 survey covariates (*water temperature*, *date of last precipitation*, *Julian date* and *Julian date*^{2}).
173 One level of our model assumed a "true" (but only partially observed) presence-absence matrix

z_{ij} for species i = 1, 2,…, N at site j =1, 2,…,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 $z_{ij} \sim Bern(\Psi_{ij})$, and Ψ_{ij} is the probability that a 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 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*, *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

197 squared standardized score of Julian days since January 1, *Water temperature* was defined as the
198 standardized value of water temperature in degrees, and *Date of last precipitation* was defined as
199 the number of days since the last precipitation event. We included the Julian date (for linear
200 effect) and Julian date squared (for squared effects along a normal distribution) because the
201 capture probability, due to activity, may change during our sampling period from March to June.
202 Standardization of covariates allowed for the estimation of Ψ and Θ at mean site and survey
203 covariates from model-generated estimates of u_i (species-specific mean probability of
204 occurrence) and v_i (species-specific mean probability of detection). Standardization of covariates
205 also enabled direct comparison of the model coefficients as effect sizes relative to variation in
206 each covariate. Our parameters u_i and v_i followed a joint normal distribution such that $[u_i, v_i \mid \Sigma] \sim$
207 $N(0, \Sigma)$ (Dorazio et al., 2006), where Σ denotes a 2 x 2 symmetric matrix with diagonal elements
208 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
209 covariance in u_i and v_i (Dorazio and Royle, 2005).

210 Seven species-specific parameters were estimated by the model (u_i, $\alpha 1_i$, v_i, $\beta 1_i$, $\beta 2_i$, $\beta 3_i$,
211 $\beta 4_i$). Community summaries (μ) were estimated by another hierarchical level of the model
212 assuming that the species-specific parameters were random effects, each governed by a
213 community-level hyper-parameter. For example, $\alpha 1_i \sim N(\mu_{\alpha 1}, \sigma_{\alpha 1})$ where $\mu_{\alpha 1}$ is the mean
214 community response (across all species) to the *MTR/VF* covariate ($\alpha 1$), and $\sigma_{\alpha 1}$ is the standard
215 deviation in $\alpha 1$ across species (Kèry et al., 2009). Using this hierarchical method, estimation of
216 species-specific parameters can be precise, even where species are rare (Zipkin et al., 2009).

217 Our model used uninformative priors for the hyper-parameters and community
218 summaries (e.g., U(0,5) for all σ parameters and U(-10 to 10) for μ_α and μ_β parameters). We
219 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 were 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.

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

242 attributes, stationary distribution appeared to be achieved based on well-mixed history plots and
243 the Gelman and Rubin statistic (<1.001 for all monitored parameters; Gelman and Rubin, 1992).
244 We detected 9 salamander species during our active searches; raw counts of salamander
245 species at control sampling transects ranged from 2 to 6, species counts at MTR/VF sampling
246 transects ranged from 0 to 4. However, we only considered 5 species (i.e., *D. fuscus*, *D.*
247 *monticola*, *E. cirrigera*, *G. porphyriticus*, and *P. ruber*) in our analysis as these species are
248 primarily associated with streams. We detected a total of 97 salamanders at MTR/VF sites and
249 804 salamanders at control sites. Some species were rarely detected at MTR/VF sites; for
250 example, only two *G. porphyritcus* and five *P.ruber* individuals were detected at MTR/VF
251 streams. Mean baseline species detection probabilities ranged from 0.38 (95% CI 0.16-0.63) for
252 *P. ruber* to 0.72 (95% CI 0.58-0.85) for *G. porphyriticus.* Model estimated detection parameters
253 were not strongly influenced by sampling covariates.
254 Our model indicated high rates of mean species occupancy across all 5 species at control
255 streams; mean estimated occupancy probabilities ranged from 0.73 (95% CI 0.41-0.96) for *P.*
256 *ruber* to 0.90 (95% CI 0.77-0.98) for *E. cirrigera* (Fig. 1). Occupancy was lower at MTR/VF
257 streams, with mean estimated occupancy probability ranging from 0.23 (95% CI 0.04-0.51) for
258 *G. porphyriticus*, to 0.62 (95% CI 0.36-0.86) for *E. cirrigera* (Fig. 1). Despite having high
259 posterior standard errors, we found that the species-specific $\alpha 1_i$ parameter estimates were all
260 negative and 95% credible intervals did not overlap with zero in any case, which collectively
261 indicates that all species were less likely to occupy MTR/VF streams (i.e., *D. fuscus*, -2.41 (95%
262 CI -3.96 – (-0.27), *D. monitcola*, -2.42 (95% CI -4.13– (-0.70), *E.cirrigera*, -2.20 (95% CI -3.96
263 – (-0.39), *G. porphyriticus*, -3.70 (95% CI -4.93 – (-2.06), *P. ruber*, -2.68 (95% CI -4.54 (95%
264 CI -4.13– (-0.79)). For our model, stationary distributions appeared to be achieved based on

265 well-mixed history plots and the Gelman and Rubin statistic (<1.001 for all monitored
266 parameters; Gelman and Rubin, 1992).
267 When all the salamander species were considered together, as a community the mean
268 occupancy in MTR/VF streams was 0.50 (95% CI 0.06-0.95) and mean occupancy in control
269 streams was 0.87 (95% CI 0.64-0.96) suggesting that salamanders have a higher probability of
270 occupancy in streams that have not been affected by MTR/VF. The 95% credible interval for the
271 occupancy covariate (μ_{a1} MTR/VF) contained only negative values -1.94 (95% CI -3.31-(-0.31))
272 and the 95% credible interval for the standard deviation (i.e., 1.20 (95% CI 0.090-3.15), in the
273 response to the covariate across species (σ_{a1} MTR/VF) was less than the absolute value of the
274 mean estimate, indicating certainty in the mean response across species (Table 2). All of the
275 mean parameter estimates for detection covariates ($\mu_{\beta 1}$ – Julian date, $\mu_{\beta 2}$ – Julian date squared,
276 $\mu_{\beta 3}$ – Water temperature, and $\mu_{\beta 4}$ – Date of last precipitation) covered zero and contained both
277 positive and negative values in the 95% credible intervals, indicating uncertainty in the mean
278 community responses to these covariates (Table 2). The mean species richness estimate for
279 MTR/VF streams was 2.27 (±1.27 SD) whereas richness was 4.67 (± 0.65 SD) for control
280 streams (mean difference of 2.29 [95% CI 1.97-2.65)] between control and MTR/VF).

281 **Discussion**

282 We found that streams impacted by MTR/VF had reduced salamander occupancy and
283 species richness and altered environmental attributes compared to control streams. Recent
284 research in West Virginia found that stream salamander abundance was reduced in first and
285 second-order MTR/VF streams compared to reference streams, yet species richness did not differ
286 between MTR/VF streams and control streams (Wood and Williams, 2013). Based on our
287 analysis, mean occupancy rates across five stream salamander species were reduced in MTR/VF

288 compared to control streams. We recognize, however, that local abundance might be a source of
289 systematic, detection bias in our study (See Royle and Nichols 2003). If detectability is
290 dependent on abundance then our model will not be able to separate sites with very low
291 abundance from those unoccupied by salamanders. However, given the assumption of equal
292 detectability between site types in our current analysis we conclude that species occupancy and
293 species richness differ between stream types.

294 Reduced salamander occupancy and species richness may be due to a complex set of
295 interacting factors operating in both terrestrial and aquatic habitats. First, the deposition of
296 overburden into valleys results in the permanent loss or burial of most of the length of low-order
297 streams within valleys (Palmer et al., 2010). The permanent loss of streams likely reduces
298 connectivity among salamander populations across landscapes, leading to reduced gene flow and
299 possible local extinction for some species (i.e., Munshi-South et al., 2013). Second, MTR/VF
300 streams often have reduced forest cover within catchments, which has been shown to be
301 negatively correlated with salamander occupancy rates and abundances (i.e., Ford et al., 2002;
302 Price et al., 2011; Price et al., 2012). Indeed, the MTR/VF streams had, on average, 75% less
303 forest cover than control streams; land-cover within MTR/VF catchments was dominated by
304 non-native grasses and shrubs. Reduction of forest cover within stream catchments may be
305 particularly detrimental to species such as *P. ruber* and *E. cirrigera*, which extensively use
306 terrestrial habitats during the non-breeding season (Petranka 1998). Additionally, Wood and
307 Williams (2013) noted lower terrestrial salamander abundance and species richness within
308 reclaimed, grass-dominated surface mine and suggest that poor soils, reduced vertical structure
309 of vegetation, little tree cover, and inadequate litter and wood debris cover contributed to their
310 findings. Nature history differences, such as use of terrestrial habitat by stream salamanders,

311 may contribute interspecific differences we observed in occupancy probability; however formal
312 tests are needed to determine relationships between salamander natural-history traits and
313 sensitivity to MTR/VF.

314 Land-cover changes on MTR/VF sites lead to numerous changes in hydrology and
315 alterations to in-stream habitat, which may also lead to decreased salamander occupancy and
316 species richness. Reclaimed mined sites have soils containing un-weathered rock that is heavily
317 compacted to reduce erosion, altered water tables, and disturbed flow paths (Bonta et al., 1992;
318 Bernhardt and Palmer, 2011). In particular, compacted soils lead to high rates of storm water
319 runoff; Negley and Eshleman (2006) and Ferrari et al. (2009) found that MTR/VF streams had
320 tripled storm runoff and doubled flow rates compared to reference catchments. High peak flows
321 have been shown to negatively affect survival of larval *E. cirrigera* in urban settings (Barrett et
322 al., 2010) and may influence survival and occupancy rates within MTR/VF streams. Altered
323 hydrology is often apparent through stream bank erosion and sedimentation, which can be
324 excessive in MTR/VF streams (Fox, 2009). Sedimentation results in burial of rocks and boulders
325 and the infilling of interstitial spaces between rocks, which reduces available microhabitats for
326 salamanders (Lowe and Bolger, 2002). Wood and Williams (2013) suggest that sedimentation
327 contributed to lower abundances of stream salamanders in West Virginia MTR/VF streams and
328 Redmond (1980) found Black Mountain Dusky Salamanders (*D. welteri*) were excluded from
329 highly silted streams due to coal mining.

330 We found MTR/VF streams had elevated levels of specific conductance, sulfates, total
331 organic carbon, and dissolved ion concentrations. A previous study conducted at the Laurel Fork
332 mine also found elevated specific conductance levels and dissolved ion concentrations at three of
333 our study sites (Fritz et al., 2010), and numerous investigations on the effects of MTR/VF on

334 water chemistry corroborate our results (i.e., Hartman et al., 2005; Pond et al., 2008; Wood and
335 Williams, 2013). Amphibians are poor osmoregulators; high specific conductance has been
336 shown to have a wide range of adverse effects (i.e., physical abnormalities, reduced survivorship,
337 reduced activity, increased corticosterone levels) on larval stages of amphibians (Sanzo and
338 Hecnar, 2006; Karracker, 2008; Chambers, 2011), perhaps resulting in population declines and
339 species extirpations. Miller et al. (2007) found that larval *E. cirrigera* abundance was negatively
340 related to specific conductance levels in urban streams and Schorr et al. (2013) found that
341 occurrences of four salamander species of the Cumberland Plateau (*D. fuscus, P. ruber, E.*
342 *cirrigera, G. porphyriticus*) were negatively correlated with elevated specific conductance levels
343 (i.e., >100 μS/cm). Stream invertebrates are an important prey item for salamanders (Petranka,
344 1998; Davic and Welsh, 2004) and decreases in macroinvertebrate populations due to water
345 chemistry are well documented in streams impacted by MTR/VF (Pond, 2010,2012). Thus,
346 adverse effects on larval amphibians combined with a reduction in prey items may lead to
347 decreases in salamander occupancy and species richness.
348 The disturbance caused by MTR/VF is drastically changing the central Appalachian
349 landscape, compromising the natural ecological and functional state of both terrestrial and
350 aquatic environments. The reclamation process, emphasizing soil compaction and the
351 establishment of non-native herbaceous species, has hindered the establishment of native tree
352 species on MTR sites (Zipper et al., 2011). These terrestrial impacts in combination with the VF
353 influences stream ecosystems. Simmons et al. (2008) suggests that MTR/VF leads to long-
354 lasting changes to terrestrial and aquatic ecosystem function; in fact, full recovery of species
355 diversity in streams impacted by MTR/VF has not been documented (Palmer et al. 2010).
356 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 (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). 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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512 **Table 1.** Mean, 95% credible intervals (95% CI), and differences in environmental attributes at Mountain-top removal/valley fill and

513 control (i.e., forest) intermittent streams located in the interior rugged section of the Cumberland Plateau in Breathitt and Knott

514 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 Conductance (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

515

516 **Table 2.** Summary of hyper-parameters for occupancy and detection covariates for salamanders

517 observed at sites of mountaintop removal and natural second growth forest streams (controls)

518 located in the interior rugged section of the Cumberland Plateau, Kentucky.

Community level hyper-parameter		Mean	Standard Deviation	95% Credible Interval	
*$\mu_{\alpha 1}$	MTR/VF	-1.97	0.75	-3.31	-0.31
*$\sigma_{\alpha 1}$	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

519 * The symbol μ indicates mean community response, while σ indicates the standard deviation in

520 the response to the covariate across species.

521 **Figure Legends**

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

526