

1 **Effects of mountaintop removal mining and valley fills on occupancy and abundance of**  
2 **stream salamanders**

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24 **Summary.**

25 **1.** Across central Appalachia (USA), mountaintop removal mining and valley filling (MTR/VF)  
26 represents a stressor to stream ecosystems. Salamanders are the dominant vertebrate in  
27 Appalachian low-order streams, yet very few studies have addressed the effects of MTR/VF on  
28 their populations. Thus, we addressed the basic question: is salamander occupancy and  
29 abundance given occupancy (hereafter conditional abundance) reduced in streams impacted by  
30 MTR/VF compared to control streams?

31 **2.** We conducted repeated counts of adult and larva of five salamander species within 10 m  
32 reaches in 11 valley-filled streams on reclaimed mined land and 12 control streams in  
33 southeastern Kentucky (USA). Physical habitat and water chemistry variables were also recorded  
34 for each stream. Relationships between salamander occupancy, conditional abundance, and site  
35 type (MTR/VF vs. control) were modeled using the hurdle model (Dorazio et al. (2013)), an  
36 extension of the binomial mixture model, where occupancy is modeled separately from  
37 abundance while accounting for differences in per-individual detection probabilities among  
38 species.

39 **3.** Using a Bayesian framework, we found mean occupancy probabilities were  $>0.84$  for all  
40 species and life stages in control reaches, whereas mean occupancy probabilities were relatively  
41 lower at reaches impacted by MTR/VF (ranging from 0.23 to 0.66). Posterior means of the  
42 difference in occupancy between site types was negative across all five species and stages,  
43 although MTR/VF stream reaches were at least 95% less likely to be occupied by *Gyrinophilus*  
44 *porphyriticus* (spring salamander), adult *Eurycea cirrigera* (southern two-lined salamander) and  
45 *Desmognathus* larvae (larval dusky salamanders) compared to control stream reaches.

46 4. Across all species and stages, mean conditional abundance was relatively lower at MTR/VF  
47 reaches compared to control reaches. Posterior means of the difference in conditional abundance  
48 between MTR/VF and control stream reaches were negative across all species and stages; 95%  
49 credible interval for difference in conditional abundance covered zero for only one species  
50 (*Pseudotriton ruber*; red salamander). After adjusting for goodness-of-fit, point estimates of  
51 differences in occupancy and conditional abundance still remained below zero for most species,  
52 reinforcing our initial findings. Additionally, we found that MTR/VF reaches had higher ion  
53 concentrations, total organic carbon, pH and specific conductance compared to control stream  
54 reaches

55 4. *Synthesis and applications.* Our study is among the first to conclude that streams impacted by  
56 MTR/VF have reduced salamander occupancy and conditional abundance compared to control  
57 streams, with responses relatively similar among species and life stages. Although the precise  
58 stressors (i.e., chemical, hydrologic, catchment alterations, etc.) responsible for reduced  
59 salamander occupancy and abundance remain elusive, our study adds to the growing literature  
60 that suggests that current reclamation and mitigation approaches do not offset the impacts of  
61 MTR/VF on populations of stream biota and ecosystems.

62

### 63 ***Introduction.***

64 Mountaintop removal mining, a form of surface mining that involves the removal of  
65 significant amounts of rock to access shallow coal seams, is a well-known stressor to ecosystems  
66 throughout the Central Appalachia region of the eastern USA (Bernhardt & Palmer 2011;  
67 Wickham *et al.* 2013). In particular, mountaintop removal mining often results in the complete  
68 or partial burial of low-order streams via valley filling, the process by which rock (i.e.,

69 overburden materials) is discarded from the mine site into adjacent valleys. Valley-filling alters  
70 stream water chemistry; leaching and surface runoff from the unweathered overburden materials  
71 leads to increased specific conductance, elevated levels of total dissolved solids, sulfates,  
72 chlorides and other ions and altered pH compared to reference streams (Fritz et al., 2010; Palmer  
73 et al., 2010; and Lindberg et al., 2011; Griffith et al., 2012). Alterations to stream catchments  
74 from mining, including forest removal and soil compaction, result in larger storm runoff  
75 coefficients, greater total runoff, higher peak hourly runoff rates, and increases in sedimentation  
76 compared to reference streams (Wiley et al., 2001; Negley & Eshleman, 2006; Simmons et al.,  
77 2008; Ferrari et al., 2009). Although laws exist to regulate the environmental impacts of  
78 mountaintop removal mining (e.g., Surface Mining Control and Reclamation Act (SMCRA) of  
79 1977; sections 402 and 404 of the Clean Water Act) through restoration and reclamation of  
80 terrestrial environments prior to abandonment and mitigation of aquatic systems, mountaintop  
81 removal mining and valley filling (MTR/VF) consistently results in long-term changes to stream  
82 ecosystem function via impacts to stream catchments, riparian zones and aquatic environments  
83 (Simmons et al., 2008).

84 Mountaintop removal mining is widespread in the central Appalachian states of Kentucky,  
85 Tennessee, Virginia, and West Virginia, a region known for high levels of stream biodiversity  
86 (Bernhardt & Palmer, 2011). Salamanders represent the dominant vertebrates in low-order  
87 streams in Appalachia where they can reach exceptional population densities (i.e., Peterman,  
88 Crawford & Semlitsch, 2008) and drive numerous ecosystem-level processes. For example,  
89 salamanders convert large quantities of invertebrate prey to vertebrate biomass; they provide a  
90 food resource to mammals, birds, fish and reptiles (Davic, 1983; Petranka, 1998; Davic &  
91 Welsh, 2004) and influence nutrient cycling and energy flow within streams and from aquatic to

92 terrestrial environments (Davic & Welsh 2004; Greene, Lowe & Likens, 2008; Keitzer &  
93 Goforth 2012). It is well documented that salamander populations are sensitive to land-use  
94 changes, such as urbanization (Barrett & Guyer, 2008; Price et al., 2011; Price, Browne, &  
95 Dorcas, 2012) and timber harvest (Lowe & Bolger, 2002; Crawford & Semlitsch, 2007; Moseley  
96 et al., 2008), that impact streams and their catchments. Despite the ubiquity of MTR/VF in  
97 central Appalachia, only Wood and Williams (2013) examined the effects of MTR/VF on  
98 relative abundances (i.e., total number of individuals counted across all species) of adults and  
99 larva; they found that relative abundance was greater in three reference streams compared to  
100 three streams impacted by MTR/VF. However, counts of some species (i.e., *Desmognathus*  
101 *fuscus* (northern dusky salamander)) were similar between MTR/VF streams and reference  
102 streams (Wood & Williams, 2013). Thus, examinations that focus on species- and/or stage-  
103 specific responses may be necessary to fully elucidate the effects of MTR/VF on salamander  
104 populations.

105 In this study, we evaluated the effects of MTR/VF on stream salamanders in the eastern  
106 Kentucky coalfields. Specifically, we modeled counts of adult and larva of five salamander  
107 species using binomial mixture models assuming a zero-inflated Poisson distribution of  
108 abundance at each stream. The zero-inflated Poisson model allows for occupancy of the site to  
109 be modeled separately from the abundance, given the site was occupied, while accounting for  
110 variation in per-individual detection probability (e.g., Dorazio et al. 2013; Wenger & Freeman,  
111 2008). We ask the basic question: is salamander occupancy and abundance, given occupancy  
112 (hereafter conditional abundance), reduced in stream reaches impacted by MTR/VF compared to  
113 control stream reaches? In addition, we report water chemistry and other physical

114 characteristics of valley-filled streams and control streams to determine stressors potentially  
115 responsible for patterns of species' occurrence and conditional abundance.

116 ***Methods.***

117 ***Study Sites.***

118 Our study sites consisted of 23 low-order headwater streams located in the interior  
119 rugged section of the Cumberland Plateau in Breathitt and Knott Counties, Kentucky USA.  
120 Specifically, we sampled stream salamanders at 11 streams, partially buried by overburden (i.e.,  
121 valley-filled), located on the reclaimed Laurel Fork surface mine (4144091.438 N 307635.435 E  
122 Zone 17) and 12 control streams in approximately 80 year-old, second growth forest on the  
123 University of Kentucky's Robinson Forest. Robinson Forest is located directly northeast of the  
124 Laurel Fork surface mine; Laurel Fork surface mine was part of the Robinson Forest up until the  
125 early 1990s when mineral rights of the property were sold by the University of Kentucky. Thus,  
126 the valley-filled streams and control streams shared many attributes (i.e., similar vegetation  
127 composition and age, elevation, subsurface geology) prior to surface mining. Laurel Fork was  
128 surface mined from the late 1990s to the early 2000s, and released from bond in November 2007  
129 suggesting reclamation was satisfied.

130 Dominant vegetation within stream catchments on the Laurel Fork surface mine consisted  
131 of nitrogen-fixing herb *Sericea lespedeza* (*Lespedeza cuneata*) and grasses (tall fescue;  
132 *Schedonorus arundinaceus*), with autumn olive (*Elaeagnus umbellate*), Virginia pine (*Pinus*  
133 *virginiana*), white oak (*Quercus alba*) and black locust (*Robinia pseudoacacia*) scattered  
134 throughout the landscape (See Fritz et al., 2010 for additional details). Although average percent  
135 forest cover within stream catchments was low at study sites on the Laurel Fork surface mine  
136 (25.0%) forest cover, riparian zones and adjacent terrestrial habitat was primarily forested (See

137 Muncy et al., 2014). Control streams were located on the main block of Robinson Forest and  
138 were dominated by second growth, mixed, mesophytic forests (i.e., white oak (*Q. alba*), tulip  
139 tree (*Liriodendron tulipifera*), eastern Hemlock (*Tsuga canadensis*) (See Phillippi & Boebinger,  
140 1986). Average forest cover within control stream catchments exceeded 99% (See Muncy et al.,  
141 2014). The average catchment sizes were similar between stream types (control stream = 24.70  
142 ha ( $\pm 21.34$  SD); MTR/VF stream = 24.51 ha ( $\pm 15.48$  SD)) (See Muncy et al., 2014).

### 143 ***Study Species.***

144 We focused on estimating the occupancy and conditional abundances of five species of  
145 stream salamander. All were lungless salamanders in the Family Plethodontidae that require low-  
146 order streams for egg deposition and larval development. *Desmognathus fuscus* and *D. monticola*  
147 (seal salamander) are streamside species, usually found within a few meters from the stream  
148 bank or under rocks and logs within the stream itself (Barbour et al., 1969). Although adult *D.*  
149 *monticola* are often larger than adult *D. fuscus*, larval period for both species lasts 8-9 months  
150 (Juterbock, 1984). *Eurycea cirrigera* (southern two lined salamander), a slender-bodied species  
151 with a 1-2 year larval period, is well known for occupying riparian and upland environments  
152 during the non-breeding season (i.e., summer and fall) (Petranka, 1998). *Gyrinophilus*  
153 *porphyriticus* (spring salamander) are large, heavy-bodied salamanders that are primarily found  
154 in streams, seeps, and springs (Petranka, 1998). The larval period of *G. porphyriticus* is  
155 between 3-4 years (Bruce, 1980) and they are well known predators of other salamanders  
156 (Petranka, 1998). *Pseudotriton ruber* (red salamander) also has a lengthy larval period (1.5-3.5  
157 years; Bruce, 1972); adult *P. ruber* are occasionally observed in upland forests far from water  
158 (Petranka, 1998).

### 159 ***Data collection.***

160 We conducted repeated counts of salamanders within a 10-m stream reaches in each  
161 MTR/VF and control stream. Reaches were haphazardly chosen based on their similarity of  
162 width, depth and water flow and all reaches included a pool, run and riffle section (See Muncy et  
163 al., 2014). Selecting sampling sites in this manner ensured instream habitat would be  
164 appropriate for salamanders. Most valley-filled stream reaches were located at the base of the  
165 valley fill. Despite the similarities in width and depth among sampling reaches (See Muncy et  
166 al., 2014), the number of cover objects (rocks > 50 mm in diameter and logs) within each reach  
167 were greater at control streams (48 cover objects) than MTR/VF streams (24 cover objects; See  
168 Muncy et al., 2014).

169 We collected a 50 mL water sample from each stream reach prior to each sampling event  
170 to evaluate the effects of MTR/VF on stream water chemistry. In the laboratory, the water  
171 samples were analyzed for concentrations of calcium (Ca), magnesium (Mg), potassium (K),  
172 sodium (Na), sulfates ( $\text{SO}_4^{2-}$ ), total organic carbon (TOC), pH and specific conductance ( $\mu\text{S}$ );  
173 sampling and analyses were performed in accordance with standard methods (Greenberg,  
174 Clesceri & Eaton, 1992). These data have been formally analyzed in Muncy et al. (2014),  
175 although we present site-specific averages and standard errors in Table 1.

176 Salamander count data were obtained through area-constrained active searches,  
177 specifically dipnetting and stream bank searches within the 10 m reach (See Price, Browne, &  
178 Dorcas, 2012). Dipnetting consisted of one sampler, moving from downstream to upstream,  
179 actively netting for adult and larval salamanders around and under cover objects (i.e., logs,  
180 rocks) and leaf litter within the stream. Stream bank searches also were conducted by one person;  
181 these searches included close examination of cover objects and other material within 1-m of the  
182 wetted width of the stream reach. Active searches were conducted during daylight hours and in



183 base flow conditions. In general, dipnetting and bank searches took 30 and 15 minutes,  
184 respectively. After searches were complete, all salamanders were identified to species, life stage  
185 (larva or adult), counted and released back into the 10-m reach. We sampled each 10-m reach  
186 four times (i.e., usually monthly) from March through June 2013.

187 We also recorded several sampling covariates at each stream reach prior to each active  
188 search to account for the conditions that may influence our ability to detect salamanders. We  
189 assumed that per-individual detection rate (i.e., probability of detecting an individual that in  
190 present in a reach and available for detection) of salamanders may be influenced by three  
191 covariates: water temperature (C°), number of days since last precipitation event, and date (i.e.,  
192 day of year). Both water temperature and number of days since last precipitation have been  
193 suggested as important predictors of activity in stream salamanders (Orser & Shure, 1975;  
194 Johnson & Goldberg, 1975; Price, Browne, & Dorcas, 2012). We recorded date because  
195 activity, and potentially detection, may change over the four months of sampling.

### 196 *Statistical Analysis.*

197 To examine the effects of mountaintop removal mining on stream salamander occupancy  
198 and conditional abundance, we modeled counts of salamander species and life stages using the  
199 extended binomial mixture model of Dorazio et al. (2013). This model assumes an underlying  
200 zero-inflated Poisson distribution of the population size at each stream which Dorazio et al.  
201 (2013) call the hurdle model. The hurdle model is parameterized so that occupancy of the  
202 streams is modeled separately from the abundance among occupied locations (i.e., conditional  
203 abundance), while still accounting for differences in per-individual detection probabilities among  
204 species. Assumptions of this model are that the population at each stream is closed to entry (i.e.,  
205 birth, immigration) and exit (i.e., death, emigration). [Simon Bon1] This model was appropriate

206 for our salamander repeated count data because it accounts for the excess of zeros in observed  
 207 counts in the MTR/VF stream reaches; failure to account ~~foref~~ this variation can substantially  
 208 bias abundance estimates (See Dorazio et al. 2013). Salamander counts on the four visits were  
 209 assumed to follow independent binomial distributions ~~conditional~~[Simon Bon2] on the site-  
 210 specific population size. Detection probabilities were allowed to depend on environmental  
 211 covariates but were otherwise assumed to be the same for MTR/VF and control streams.

212 To conduct our analysis, we first separated salamander count data by species and in some  
 213 instances stage (i.e., adult vs. larva) for each of the five species. Specifically, we separately  
 214 analysed counts for only adult *D. fuscus*, *D. monticola* and *E. cirrigera*. We combined counts of  
 215 adult and larval forms of *G. porphyriticus* as well as counts of *P. ruber*, as all but one of the  
 216 captures of these two species were larvae. Finally, we analysed larval counts of *E. cirrigera* and  
 217 combined counts for larval *D. fuscus* and *D. monticola* due to difficulty of identifyingseparating  
 218 *Desmognathus* species in larval form; collectively we refer to these as *Desmognathus* larvae.

219 We first modelled occupancy of species or life stage  $s$  at site  $I$ ,  $O_{si}$ , as a Bernoulli random  
 220 variable (e.g., MacKenzie et al. 2003) where the probability that site  $i$  was occupied by species or  
 221 life stage  $s$ ,  $\psi_{si}$ , was allowed to vary between MTR/VF ( $m_i = 1$ ) and control stream reaches ( $m_i =$   
 222 0) modelled on the logistic scale, given by:

$$223 \quad \text{logit}(\psi_{si}) = \begin{cases} \gamma_{s1} & m_i = 0 \\ \gamma_{s2} & m_i = 1 \end{cases} . \quad (1)$$

224 The difference in  $\psi_{si}$  between MTR/VF and control stream reaches for species or life-stage  $s$  was  
 225 measured through a pairwise contrast such that  $\Delta\gamma_s = \gamma_{s1} - \gamma_{s2}$ . Differences in occupancy  
 226 between MTR/VF and control streams ~~waswere~~ assessed by considering whether or not the ~~(1-~~  
 227 ~~a)100%~~Central 50 and 95% credible intervals for  $\Delta\gamma_s$  covered zero ~~for different values of a-~~  
 228 ~~Specifically, we focused on the coverage of the central 50% and 95% credible intervals.~~

229 Abundance for each species and life stage  $s$  at site  $i$  ( $N_{si}$ ) was then modelled conditional on  
 230 occupancy. If the site was unoccupied by a given species or life stage  $s$ , we considered the  
 231 abundance to be zero. Otherwise, the abundance at site  $i$  was modelled as a zero truncated  
 232 Poisson random variable with rate parameter  $\lambda_{si}$  such that:

$$233 \quad P(N_{si} = n | O_{si} = 1) = \frac{\lambda_{si}^n}{(e^{\lambda_{si}} - 1)n!}, \quad n = 1, 2, 3, \dots \quad (2)$$

234 Conditional on site  $i$  being occupied, the expected abundance was:

$$235 \quad E(N_{si} | O_{si}) = \frac{\lambda_{si} e^{\lambda_{si}}}{e^{\lambda_{si}} - 1} \quad (3)$$

236 The distribution of abundance given occupancy was allowed to vary between control and  
 237 MTR/VF streams and among species ~~and~~/or life stages so that:

$$238 \quad \lambda_{si} = e^{(m_i - 1)\beta_{s1} + m_i \beta_{s2}} \quad (4)$$

239 The difference in abundance given occupancy between MTR/VF and control streams for species  
 240 or life-stage  $s$  was measured as  $\Delta B_s = B_{s1} - B_{s2}$ ; differences were assessed by considering  
 241 whether or not the ~~(1- $\alpha$ )100%~~50 and 95% credible ~~intervals~~interval for  $\Delta B_s$  covered zero ~~for~~  
 242 ~~different values of  $\alpha$ . Again, we focused on the 50% and 95% credible intervals.~~

243 Accounting for both occupancy and abundance given occupancy, the distribution for the  
 244 overall abundance of species or life-stage  $s$  at site  $i$  was given by the probabilities

$$245 \quad P(N_{si} = n) = \begin{cases} (1 - \psi_{si}) & n = 0 \\ \frac{\psi_{si} \lambda_{si}^n}{n!(e^{\lambda_{si}} - 1)} & n = 1, 2, 3, \dots \end{cases} \quad (5)$$

246 ~~Provided that  $(1 - \psi_{si}) > e^{-\lambda_{si}}$ ,  $\psi_{si} > 0$~~  This represents a zero-inflated Poisson distribution in which  
 247 occupancy ~~may be is~~ less than expected under a strict Poisson model for abundance. The  
 248 expected abundance of species or life stage  $s$  at site  $i$  (i.e.,  $E(N_{si})$ ) combining the models for both  
 249 occupancy and abundance given occupancy was

250 
$$E(N_{si}) = \frac{\psi_{si} \lambda_{si} e^{\lambda_{si}}}{e^{\lambda_{si}} - 1} \quad (6)$$

251 Differences in the overall abundance at MTR/VF and control stream reaches was assessed by  
 252 comparing  $E(N_{si})$  and  $E(N_{sj})$  where site  $i$  was mined ( $m_i = 1$ ) and site  $j$  was a control site ( $m_j = 0$ ).

253 Individual [Simon Bon3] salamanders were assumed to behave independently within and  
 254 between sampling occasions so the number of captures of species/life stage  $s$  at site  $j$  on occasion  
 255  $i$  followed a binomial distribution

256 
$$Y_{sij} | N_{si} \sim \text{Binomial}(N_{si}, p_{sij}) \quad (7)$$

257 The probability of capture was modeled on the logit scale as:

258 
$$\text{logit}(p_{sij}) = \alpha'_s x_{sij} \quad (8)$$

259 where  $x_{sij}$  was a vector of sampling covariates and  $\alpha_s$  the associated vector of coefficients for  
 260 species/life-stage  $s$ . We assumed that per-individual detection rate of salamanders may be  
 261 influenced by three covariates: water temperature ( $C^\circ$ ), number of days since last precipitation  
 262 event, and Julian date, which we standardized so the mean of the population was 0 and the  
 263 standard deviation was 1.

264 Dorazio et al (2013) provided maximum likelihood methods to fit the hurdle models.  
 265 However, we chose to model each species and life stage separately in a Bayesian framework  
 266 using Markov chain Monte Carlo (MCMC) sampling in OpenBUGS to generate samples from  
 267 the posterior distribution (Lunn et al., 2009). All parameters were assigned non-informative  
 268 [U4][Simon Bon5] prior distributions. Three parallel chains were run in OpenBUGS for each model  
 269 so that convergence could be assessed via the Gelman-Rubin diagnostic (Gelman & Rubin,  
 270 1992). Each chain was run for 50,000 iterations in total, the first 25,000 were removed as burn-  
 271 in, and the final 25,000 were thinned by a factor of 5. This provided a total of 15,000 samples  
 272 from which we approximated posterior summary statistics including the mean, standard

273 deviation, and [50 and 95% credible intervals](#)<sup>[U6]</sup> for each of the model parameters. Posterior  
274 summary statistics were also computed for derived quantities including the occupancy  
275 probability and abundance of each species or life stage at each of the sites.

276 Goodness-of-fit was conducted after the initial [modell](#)<sup>ing</sup> stage by computing Bayesian p-  
277 values based on the contributions to the completed data likelihood function for each species or  
278 life-stage and each site. The complete data likelihood function was constructed by treating  
279 occupancy and abundance as if they were part of the observed data. In particular, we constructed  
280 three discrepancy measures from the log of the likelihood components modeling the abundance  
281 (including occupancy) and the observed values conditional on abundance, and from the overall  
282 likelihood (i.e., See Kéry & Schaub, 2012). Bayesian p-values were then computed by  
283 comparing the discrepancy measures obtained for both the observed data and simulated data  
284 generated on each iteration of the MCMC algorithm (See Gelman, Meng & Stern, 1996). We  
285 considered Bayesian p-values  $\leq 0.10$  for any of the three discrepancy measures as indications of  
286 potential [influential points in the sensitivity analysis](#) [lack of fit](#). For each species and life-stage,  
287 the analysis (see occupancy and abundance analyses above) was repeated after removing all sites  
288 producing a Bayesian p-value  $\leq 0.10$  for any of the three discrepancy measures (See Appendix  
289 S1 Sensitivity Analysis in Supporting Information).

## 290 **Results.**

291 We counted 92 *D. fuscus* adults, 117 *D. monticola* adults, 46 *E. cirrigera* adults, 106 *G.*  
292 *porphyriticus*, 22 *P. ruber*, 190 *E. cirrigera* larvae, and 188 *Desmognathus* larvae in control  
293 stream reaches. *Gyrinophilus porphyriticus* was detected at all 12 control stream reaches, adults  
294 of *D. fuscus*, *D. monticola*, and *E. cirrigera* were detected at 11 control reaches and *P. ruber*  
295 were detected at half of the control reaches. Larval *Desmognathus* and *E. cirrigera* were detected

296 at 12 and 11 control reaches, respectively. Salamander counts in MTR/VF reaches resulted in  
297 25 adult *D. fuscus*, 19 adult *D. monticola*, 4 adult *E. cirrigera*, 2 *G. porphyriticus* larvae, 5 *P.*  
298 *ruber* larvae, 11 *E. cirrigera* larvae, and 25 *Desmognathus* larvae. Of the 11 MTR/VF reaches,  
299 adults of *D. fuscus*, *D. monticola*, and *E. cirrigera* were detected at 7, 6, 3 stream reaches,  
300 respectively. *Pseudotriton ruber* (2 stream reaches) and *G. porphyritus* (1 stream reach) were  
301 rarely detected in MTR/VF streams. Larval *Desmognathus* and *E. cirrigera* were both found at  
302 5 of 11 MTR/VF stream reaches.

303 We found differences in our species-specific estimates of occupancy between control  
304 sites and MTR/VF reaches. With the exception of *P. ruber*, estimated occupancy probabilities  
305 were  $> 0.84$  for all species and stages in controls reaches (Fig. 1). Estimated occupancy  
306 probabilities were relatively low in MTR/VF reaches, ranging from 0.23 (95% CI = 0.03 – 0.67)  
307 for *G. porphyriticus* to 0.66 (95% CI = 0.37 – 0.91) for *D. fuscus* (Fig 1). Despite the variation,  
308 posterior means of  $\Delta\gamma_s$  for all species and stages were negative, suggesting decreased occupancy  
309 in reaches impacted by MTR/VF (Fig. 2). Upper bounds of the 95% credible intervals for  $\Delta\gamma_s$   
310 were below zero for three species, *G. porphyriticus*, adult *E. cirrigera* and larval *Desmognathus*,  
311 ~~indicating that MTR/VF reaches were at least 95% less likely to be occupied by these species or~~  
312 ~~life stages~~. Although the upper bounds of the 50% credible intervals for *D. fuscus*, *D. monticola*,  
313 larval *E. cirrigera* and *P. ruber* were well below zero, we could not draw strong conclusions  
314 about differences in occupancy for these species because the 95% credible intervals covered  
315 zero.

316 Our estimates of conditional abundance also exhibited both interspecific and site type  
317 differences (Fig 3). In control reaches, estimated mean conditional abundance ranged from  
318 57.43 (95% CI = 11.55-232.10) *G. porphyriticus* to 8.39 (95% CI = 4.34-17.02) *D. fuscus* per 10

319 m reach (Fig. 3). Conversely, across all species and stages, mean conditional abundance was  
320 relatively lower in MTR/VF reaches(Fig. 3). Sampling covariates did not strongly influence  
321 detection probabilities for most species as coefficients for  $\alpha$  parameters covered zero, however  
322 *G. porphyriticus* ( $\alpha = 0.30$  (95% CI = 0.08-0.53) and adult *E. cirrigera* ( $\alpha = 0.69$  (95% CI =  
323 0.09-1.36) were detected with greater frequency as the number of days since last rain increased,  
324 larval *E. cirrigera* were detected more frequently as date increased ( $\alpha = 0.78$  (95% CI = 0.57-  
325 1.02), and adult *E. cirrigera* were detected less frequently as date increased( $\alpha = -1.88$  (95% CI =  
326 -2.85 to -1.13). The 95% credible interval for  $\Delta_{Bs}$  covered zero for only one species (*P. ruber*)  
327 and posterior means  $\Delta_{Bs}$  for all species and stages were negative, suggesting reduced abundance  
328 given occupancy in reaches impacted by MTR/VF (Fig. 2).

329 The [goodness-of-fit\[Simon Bon7\]](#) assessment for all species and life stages indicated lack of  
330 fit of the models at one or more of the study reaches. For each species and life-stage, the  
331 analysis (see occupancy and abundance analysis) was repeated after removing all reaches  
332 producing a Bayesian p-value  $\leq 0.10$  for any of the three discrepancy measures (i.e., occupancy  
333 ( $O_{si}$ ), conditional abundance ( $N_{si}$ ), and the observed counts on each occasion given abundance  
334 ( $Y_{si}$ )). In all but one case, the point estimates of  $\Delta_{\gamma_s}$  and  $\Delta_{Bs}$  remained below zero, though the  
335 estimated differences were closer to zero than in the initial analysis and credible intervals were  
336 wider, which was expected given the removal of study reaches (Fig. 4; See Fig. S1). The single  
337 exception was the point estimate of the difference in abundance given occupancy for *G.*  
338 *porphyriticus* which was above zero.

339 Control reaches had consistently lower concentrations of calcium (Ca), magnesium (Mg),  
340 potassium (K), sodium (Na), and sulfates ( $\text{SO}_4^{2-}$ ) than those recorded on MTR/VF sites (Table  
341 1). Additionally, total organic carbon (TOC) and pH tended to be elevated in MTR/VF reaches

342 (Table 1). In particular, site-specific averages of specific conductance ranged from 972.50  
343 ( $\pm 86.70$ ) to 2365.00 ( $\pm 72.40$ )  $\mu\text{S}/\text{cm}$  in MTR/VF reaches whereas, in control reaches, site-  
344 specific averages ranged from 38.80 ( $\pm 5.91$ ) to 81.25 ( $\pm 25.54$ )  $\mu\text{S}/\text{cm}$  (Table 1).

345

### 346 *Discussion.*

347 Stream salamanders are known to be sensitive to land-use changes, including both pulse-type  
348 (i.e., clearcutting) and press-type (i.e., urbanization) disturbances (Crawford & Semlitsch, 2007;  
349 Price et al. 2011; Connette & Semlitsch, 2013; Surasinghe & Baldwin, 2014), however few  
350 studies have examined their response to MTR/VF, the primary type of land use disturbance in  
351 Central Appalachia (but see Wood and Williams 2013; Muncy et al. 2014). Using the hurdle  
352 model (Dorazio et al. 2013), we found occupancy and conditional abundance of most salamander  
353 species and life stages examined were reduced in stream reaches impacted by MTR/VF. Indeed,  
354 posterior means for both  $\Delta\gamma_s$  and  $\Delta B_s$  were negative for all species and life-stages; ~~however, the~~  
355 ~~credible intervals for  $\Delta\gamma_s$  were consistently much larger than for  $\Delta B_s$ , indicating that the estimated~~  
356 ~~differences in occupancy were less precise than the estimated differences in conditional~~  
357 ~~abundance.~~ [Simon Bon8] Despite the broad credible intervals, three species and/or life-stages, *G.*  
358 *porphyriticus*, adult *E. cirrigera* and larval *Desmognathus*, were unequivocally less likely to  
359 occupy streams impacted by MTR/VF, and, with the exception of *P. ruber*, all other species and  
360 life stages had clearly reduced conditional abundances in MTR/VF stream reaches.

361 Results from the sensitivity analysis were qualitatively the same as those obtained from the  
362 original analysis for many species and life-stages (i.e., *D. fuscus* and *D. monticola* adults, *E.*  
363 *cirrigera* larva, and *P. ruber*). However, results for a few species, particularly *G. porphyriticus*,  
364 showed more sensitivity to the sites with potential lack-of-fit. Yet, these results are not surprising



365 given that *G. porphyriticus* was detected at only one MTR/VF reach which was removed during  
366 the sensitivity analysis. Similarly, results from the sensitivity analysis for *Desmognathus* larva  
367 showed the 95% credible interval for  $\Delta\gamma_s$  were covered zero. Again, this was not unexpected  
368 given that nine of the 23 reaches were excluded from the sensitivity analysis for *Desmognathus*  
369 larva, greatly reducing the amount of data. This suggests either a violation of the closure  
370 assumption or that the detection model for *Desmognathus* larva was not appropriate.

371 We acknowledge some interspecific and/or stage specific differences in response to  
372 MTR/VF. For example, both *Desmognathus* species appeared to be relatively more tolerant of  
373 MTR/VF than other species in our analysis (e.g., all *Desmognathus* species and life stages had  
374 occupancy estimates  $>0.48$  and mean abundance estimates for *D. fuscus*, *D. monticola*, and  
375 *Desmognathus* larva were 2.84, 5.18, and 3.30 individuals per 10 m sampling reach in MTR/VF  
376 streams). These results contrast those of *G. porphyriticus* which had significantly reduced mean  
377 occupancy and conditional abundance in reaches impacted by MTR/VF. These findings may be  
378 related to life histories of these animals. Surasinghe and Baldwin (2014) reported large-bodied,  
379 long-lived, late maturing salamanders, such as *G. porphyriticus*, were intolerant of riparian and  
380 stream disturbances (e.g., siltation) whereas small-bodied, early maturing, short-lived species  
381 (i.e., *E. cirrigera*, *D. fuscus*) were relatively tolerant. These species- and stage- effects may be  
382 important from a population recovery potential; yet, we emphasize that all species and stages had  
383 decreased occupancy and conditional abundance in MTR/VF reaches suggesting the general  
384 intolerance of salamanders to streams impacted by MTR/VF at our study sites.

385 This reduced occupancy and conditional abundances of salamanders in streams impacted  
386 by MTR/VF is likely due to the multitude of stressors in MTR/VF streams and adjacent riparian  
387 areas. First, stream catchments on mined land often have little forest cover due to poor soil

388 conditions following mining and/or restoration techniques (Chaney et al. 1995). For example,  
389 although revegetation of mined land is required under SMCRA, this process typically involved  
390 hydroseeding the recontoured land with grasses. Our MTR/VF stream catchments contained, on  
391 average, only 25% forest cover with the remaining land-use in grasses and shrubs (see study sites  
392 above). The reduced forest cover, along with poor soils, and low amounts of litter and woody  
393 debris, may limit populations of certain species, like *E. cirrigera* and *P. ruber*, that inhabit  
394 forests during the nonbreeding season (Petranka, 1998). Secondly, compacted soils in stream  
395 catchments on MTR/VF result in increased storm runoff and stream flow rates (Negley &  
396 Eshleman, 2006; Ferrari et al., 2009; Fox, 2009), which has been linked to reduced survival of  
397 larval *E. cirrigera* (Barrett et al., 2010) and reduced recruitment in *G. porphyriticus* (Lowe &  
398 McPeck, 2012). Additionally, MTR/VF streams often have highly embedded or armored  
399 substrate, which has been connected to reduced abundances of various salamander species  
400 (Redmond, 1980; Lowe, Nislow & Bolger, 2004; Wood & Williams, 2013). The reduction in the  
401 number of cover objects within our MTR/VF reaches compared to control reaches may have  
402 been due to the increased siltation or embeddness due to mining activities.

403 Our analysis of water chemistry indicated consistently greater concentrations of ions and  
404 elevated specific conductance in MTR/VF reaches compared to control reaches. Specifically,  
405 average of specific conductance ranged from 972.50 to 2365.00  $\mu\text{S}/\text{cm}$  in MTR/VF reaches.  
406 Elevated specific conductance (i.e.,  $> 500 \mu\text{S}/\text{cm}$ ) within streams has been positively correlated  
407 in decreased macroinvertebrate abundance (Pond 2008); indeed, Pond (2010 & 2012)  
408 documented reduced abundances of mayfly taxa (Order Ephemeroptera) and many stonefly  
409 (Order Plecoptera) and caddisfly (Order Tricoptera) genera in streams impacted by surface  
410 mining, with some taxa becoming absent. Macroinvertebrates represents important prey items

411 to salamanders (Keitzer and Goforth 2013) and may be a possible explanation for decreased  
412 occupancy and conditional abundance in streams impacted by MTR/VF. Increased levels of  
413 specific conductance has been shown to influences amphibian behavior (Karraker et al. 2008),  
414 corticosterone levels and feeding (Chambers 2011). Embryonic and larval survival of amphibians  
415 has been found to be reduced at moderate (500  $\mu$ S) and high specific conductivities (3000  $\mu$ S)  
416 (Karraker et al. 2008); although we are not aware of studies conducted on Plethodontid  
417 salamanders. These aforementioned stressors and potentially others represent potential  
418 mechanisms responsible for relatively low in conditional abundance and occupancy in our  
419 MTR/VF reaches.

#### 420 ***Management Recommendations***

421 Mountaintop removal mining has impacted thousands of km of streams throughout the  
422 eastern coal fields of Kentucky, Tennessee, West Virginia and elsewhere (Barton, 2011;  
423 Bernhardt et al., 2012). Based on our results and those from other studies (i.e., Wood &  
424 Williams, 2013; Muncy et al., 2014), it appears stream salamander populations, along with  
425 macroinvertebrates (Pond 2010 & 2012) and fishes (Hopkins & Roush 2013; Hitt & Chambers  
426 (2014) have reduced levels of occupancy and abundance in streams impacted by MTR/VF.  
427 Collectively, these data strongly suggest that conventional reclamation efforts, with respect to the  
428 recovery of aquatic ecosystems, have largely been unsuccessful (Bernhardt & Palmer, 2011;  
429 Zipper et al., 2011); Palmer et al. (2010) notes that full recovery of stream biota post restoration  
430 has not been documented. Alternative reclamation and mitigation techniques, including the  
431 Forestry Reclamation Approach that advocates reforesting surface mines (See Zipper et al.  
432 2011), could be beneficial to aquatic systems and semi-aquatic organisms impacted by MTR/VF

433 by providing terrestrial habitat, restoring landscape hydrologic patterns and possibly reducing  
434 leaching of salts from mine spoils.

435

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442

443 ***References.***

444 Barton C. (2011) Coal mining versus water quality: An electrifying topic. American Water  
445 Resources Association: Water Resource Impact, 13, 23-24.

446 Barbour R.W., Hardin J.W., Schafer J.P. & Harvey M.J. (1969) Home range, movements, and  
447 activity of the dusky salamander, *Desmognathus fuscus*. Copeia, 293-297.

448 Barrett K. & Guyer C. (2008) Differential responses of amphibians and reptiles in riparian  
449 andstream habitats to land use disturbances in western Georgia, USA. Biological  
450 Conservation, 141, 2290-2300.

451 Barrett K., Helms B.S., Samoray S.T. & Guyer C. (2010) Growth patterns of a stream vertebrate  
452 differ between urban and forested catchments. Freshwater Biology, 55, 1628-1635.

453 Barrett K., & Price, S.J. (2014) Stream salamanders and urbanization: a review, conservation  
454 options and research needs. Freshwater Science 33:927-940.

455 Bernhardt E.S. & Palmer M.A. (2011) The environmental costs of mountaintop mining valley fill  
456 operations for aquatic ecosystems of the Central Appalachians. Year in Ecology and  
457 Conservation Biology. Annals of New York Academy of Science, 1223, 39-57.

458 Bernhardt E.S., Lutz B.D., King R.S., Fay J.P, Carter C.E., Helton A.M., Campagna D. & Amos  
459 J. (2012) How many mountains can we mine? Assessing the regional degradation of  
460 Central Appalachian rivers by surface coal mining. Environmental Science and  
461 Technology, 46, 8115-8122.

462 Bruce R.C. (1972) The larval life of the red salamander, *Pseudotriton ruber*. Journal of  
463 Herpetology, 6, 43-51.

464 Bruce R.C. (1980) A model of the larval period of the spring salamander, *Gyrinophilus*  
465 *porphyriticus*, based on size-frequency distributions. Herpetologica, 36, 78-86.

466 Chambers D.L. (2011) Increased conductivity affects corticosterone levels and prey consumption  
467 in larval amphibians. Journal of Herpetology, 45, 219-223.

468 Chaney, W.R, Pope, P.E., Byrnes, W.R. 1995. Tree survival and growth on land reclaimed in  
469 accord with Public Law 95-87. Journal of Environmental Quality 24:630-634.

470 Connette G.M. & Semlitsch R.D. (2013) Life history as a predictor of salamander recovery rate  
471 from timber harvest in southern Appalachian Forests, USA. Conservation Biology, 27,  
472 1399-1409.

473 Crawford J.A. & Semlitsch R.D. (2007) Estimation of core terrestrial habitat for stream-breeding  
474 salamanders and delineation of riparian buffers for protection of biodiversity.  
475 Conservation Biology, 21, 152-158.

476 Davic R.D. (1983) An investigation of salamander guild predation in a North Carolina stream: an  
477 experimental approach. PhD Thesis, Kent State University, Ohio.

478 Davic R.B. & Welsh H.H. Jr. (2004) On the ecological roles of salamanders. *Annual Review of*  
479 *Ecology, Evolution and Systematics*, 35, 405-434.

480 Dorazio, R.M., Martin, J., & Edwards, H. H. (2013) Estimating abundance while accounting for  
481 rarity, correlated behavior, and other sources of variation in counts. *Ecology*, 94, 1472-  
482 1478.

483 Ferrari J.R., Lookingbill T.R., McCormick B., Townsend P.A. & Eshleman K.N. (2009) Surface  
484 mining and reclamation effects on flood response of watersheds in the central  
485 Appalachian Plateau region. *Water Resources Research*, 45, W04407,  
486 doi:10.1029/2008WR007109

487 Fox J.F. (2009) Identification of sediment sources in forested watersheds with surface coal  
488 mining disturbance using carbon and nitrogen isotopes. *American Water Resources*  
489 *Association*, 45, 1273-1289.

490 Fritz K.M., Fulton S., Johnson B.R., Barton C.D., Jack J.D., Word D.A. & Burke R.A. (2010)  
491 Structural and functional characteristics of natural and constructed channels draining a  
492 reclaimed mountaintop removal and valley fill coal mine. *Journal of the North American*  
493 *Benthological Society*, 29, 673-689.

494 Gelman A.E., Meng X. & Stern H.S. (1996) Posterior predictive assessment of model fitness via  
495 realized discrepancies. *Statistica Sinica*, 6, 733-807.

496 Gelman A.E. & Rubin D.B. (1992) Inference from iterative simulation using multiple sequences.  
497 *Statistical Science*, 7, 457-511.

498 Greenberg A.E., Clesceri L.S. & Eaton A.D. (1992) Standard methods for the examination of  
499 water and wastewater. 18th ed. American Public Health Association, Washington, DC.

500 Greene B.T., Lowe W.H. & Likens G.E. (2008) Forest succession and prey availability influence  
501 the strength and scale of terrestrial-aquatic linkages in a headwater salamander system.  
502 *Freshwater Biology*, 53, 2234-2243.

503 Griffith M.B., Norton S.B., Alexander L.C., Pollard A.I. & LeDuc S.D. (2012) The effects of  
504 mountaintop mines and valley fills on the physicochemical quality of stream ecosystems  
505 in the central Appalachians: A review. *Science of the Total Environment*, 417, 1-12.

506 Hartman K., Kaller M., Howell J. & Sweka J. (2005) How much do valley fills influence  
507 headwater streams? *Hydrobiologia*, 532, 91-102.

508 Hitt N.P. & Chambers D.B. (2014) Temporal changes in taxonomic and functional diversity of  
509 fish assemblages downstream from mountaintop mining. *Freshwater Science*, 33, 915-  
510 926.

511 Hopkins R.L. & Roush J.C. (2013) Effects of mountaintop mining on fish distributions in central  
512 Appalachia. *Ecology of Freshwater Fish*, 22, 578-586.

513 Johnson J.E. & Goldberg A.S. (1975) Movement of larval two lined salamanders (*Eurycea*  
514 *bislineata*) in the Mill River, Massachusetts. *Copeia*, 588-589.

515 Juterbock J.E. (1984) Evidence for the recognition of specific status for *Desmognathus welteri*.  
516 *Journal of Herpetology*, 240-255.

517 Keitzer S.C. & Goforth R.R. (2013) Salamander diversity alters stream macroinvertebrate  
518 community structure. *Freshwater Biology*, 58, 2114-2125.

519 Kéry M. & Schaub M. (2012) Bayesian population analysis using WinBUGS: a hierarchical  
520 perspective. Academic Press, Waltham, Massachusetts.

521 Lemly A.D. (1997) A teratogenic deformity index for evaluating impacts of selenium on fish  
522 populations. *Ecotoxicology and Environmental Safety*, 37, 259-266.

523 Lindberg T.T., Bernhardt E.S., Bier R., Helton A.M., Merola R.B., Wengosh A. & Di Giulio  
524 R.T. (2011) Cumulative impacts of mountaintop mining on an Appalachian watershed.  
525 In: Proceedings for the National Academy of Science of the United States of America,  
526 108, 20929-20934.

527 Lowe W.H. & Bolger D.T. (2002) Local and landscape-scale predictors of salamander  
528 abundance in New Hampshire headwater streams. *Conservation Biology*, 16, 183-193.

529 Lowe W.H., Nislow K.H. & Bolger D.T. (2004) Stage-specific and interactive effects of  
530 sedimentation and trout on a headwater stream salamander. *Ecological Applications*,  
531 14,164–172.

532 Lowe W.H. & McPeck M.A. (2012) Can natural selection maintain long-distance dispersal?  
533 Insight from a stream salamander system. *Evolutionary Ecology*, 26, 11-24.

534 Lunn D., Spiegelhalter D., Thomas A. & Best, N. (2009) The BUGS project: Evolution, critique,  
535 and future directions. *Statistics in Medicine*, 28, 3049-3067.

536 MacKenzie, D.A., Nichols, J.D., Hines, J.E., Knutson, M.G. & Franklin, A.B (2003) Estimating  
537 site occupancy, colonization and local extinction when a species is detected imperfectly,  
538 *Ecology*, 84, 2200-2207.

539 Merricks T.C., Cherry D.S., Zipper C.E., Currie R.J. & Valenti T.W. (2007) Coal-mine hollow  
540 fill and settling pond influences on headwater streams in southern West Virginia, USA.  
541 *Environmental Monitoring and Assessment*, 129, 359-378.

542 Moseley K.R., Mark Ford W., Edwards J.W. & Schuler T.M. (2008) Long-term partial cutting  
543 impacts on *Desmognathus* salamander abundance in West Virginia headwater streams.  
544 *Forest ecology and management*, 254, 300-307.



545 Muncy B., Price S.J., Bonner S.J. & Barton C.D. (2014) Mountaintop removal mining reduces  
546 stream salamander occupancy and richness in southeastern Kentucky (USA). *Biological*  
547 *Conservation*, 180, 115-121.

548 Negley T.L. & Eshleman K.N. (2006) Comparison of storm-flow responses of surface-mined and  
549 forested watersheds in the Appalachian Mountains, U.S.A. *Hydrologic Processes*, 20,  
550 3467-3483.

551 Orser P.N. & Shure D.J. (1975) Population cycles and activity patterns of the Dusky salamander,  
552 *Desmognathus fuscus fuscus*. *American Midland Naturalist*, 93, 403-410.

553 Palmer M.A., Bernhardt E.S., Schlesinger W.H., Eshleman K.N., Fougoula-Georgiou E.,  
554 Hendryx M.S., Lemly A.D., Likens G.E., Loucks O.L., Power M.E., White P.S. &  
555 Wilcock P.R. (2010) Mountaintop mining consequences. *Science*, 327, 148-149.

556 Peterman W.E., Crawford J.A. & Semlitsch R.D. (2008) Productivity and significance of  
557 headwater streams: population structure and biomass of the black-bellied salamander  
558 (*Desmognathus quadramaculatus*). *Freshwater Biology*, 53, 347-357.

559 Petranka J.W. (1998) Salamanders of the United States and Canada. Smithsonian Institution  
560 Press, Washington, DC.

561 Phillippi M.A. & Boebinger A. (1986) A vegetational analysis of three small watersheds in  
562 Robinson Forest, Eastern Kentucky. *Castanea*, 51, 11-30.

563 Pond G.J., Passmore M.E., Borsuk F.A., Reynolds L. & Rose C.J. (2008) Downstream effects of  
564 mountain top coal mining: comparing biological conditions using family- and genus-  
565 level macroinvertebrate bioassessment tools. *Journal of American Benthological Society*,  
566 27, 717-737.

567 Pond G.J. (2010) Patterns of Ephemeroptera taxa loss in Appalachian headwater streams  
568 (Kentucky, USA). *Hydrobiologia*, 641, 185-201.

569 Pond G.J. (2012) Biodiversity loss in Appalachian headwater streams (Kentucky, USA):  
570 Plecoptera and Trichoptera communities. *Hydrobiologia*, 679, 97-117.

571 Price S.J., Cecala K.K., Browne R.A. & Dorcas M.E. (2011) Effects of urbanization on  
572 occupancy of stream salamanders. *Conservation Biology*, 25, 547-555.

573 Price S.J., Browne R.A. & Dorcas M.E. (2012) Evaluating the effects of urbanization on  
574 salamander abundances using a before-after control-impact design. *Freshwater Biology*,  
575 57, 193-203.

576 Redmond W.H. (1980) Notes on the distribution and ecology of the black mountain dusky  
577 salamander *Desmognathus welteri* Barbour (Amphibia: Plethodontidae) in Tennessee.  
578 *Brimleyana*, 4, 123–131.

579 Simmons J.A., Currie W.S., Eshleman K.N., Kuers K., Monteleone S., Negley J.L., Pohlrad B.R.  
580 & Thomas, C.L. (2008) Forest to reclaimed mine land use change leads to altered  
581 ecosystem structure and function. *Ecological Applications*, 18, 104-118.

582 Surasinghe, T.D. & Baldwin, R.F. (2015) Importance of riparian forest buffers in conservation of  
583 stream biodiversity: Responses to land uses by stream-associated salamanders across two  
584 southeastern temperate ecoregions. *Journal of Herpetology* 49, 83-94.

585 Wenger S.J. & Freeman M.C. (2008) Estimating species occurrence, abundance, and detection  
586 probability using zero-inflated distributions. *Ecology*, 89, 2953-2959.

587 Wickham, J., Wood, P.B., Nicholson, M.C., Jenkins, W., Druckenbrod, D., Suter, G.W., Strager,  
588 M.P., Mazzarella, C., Galloway, W. & Amos, J. (2013). The overlooked terrestrial  
589 impacts of mountaintop mining. *BioScience* 63:335-348.

- 590 Wiley J.B., Evaldi R.D., Eychaner J.H. & Chambers D.B. (2001) Reconnaissance of stream  
591 geomorphology, low streamflow, and stream temperature in the mountaintop coal-mining  
592 region, southern West Virginia, 1999-2000. US Department of the Interior, US  
593 Geological Survey.
- 594 Wood P.B. & Williams J.M. (2013) Impact of valley fills on streamside salamanders in southern  
595 West Virginia. *Journal of Herpetology*, 47, 119-125.
- 596 Zipper C.E., Burger J.A., Skousen J.G., Angel P.N., Barton C.D., Davis V. & Franklin J.A.  
597 (2011) Restoring forests and associated ecosystem services on Appalachian coal surface  
598 mines. *Environmental Management*. 47, 751-765.

***Tables.***

Table 1. Site-specific average ( $\pm$  SE) for select water chemistry attributes at mountaintop removal and valley fill (VF) and control streams (R) located in eastern Kentucky, USA.

Overall treatment means are represented by VF or R Treatment. Parameters denoted with an asterisk indicate limited sample size. One asterisk denotes one sample, while two asterisks represent two samples. Manganese means noted as 0.00 mg/L were below detection limits.

Study Site	Specific Conductance									
	$\mu\text{S/cm}$	TOC (mg/L C)	pH [H+]	Mn (mg/L)	SO4 (mg/L)	Ca (mg/L)	Mg (mg/L)	K (mg/L)	Na (mg/L)	
VF - 1	1692.00 (113.61)	10.52 (8.40)	6.39 (0.31)	2.69 (1.02)	379.04 (181.02)	28.16 (2.48)	10.39 (0.15)	8.48 (0.42)	9.14 (0.43)	
VF - 2	1644.67 (112.06)	25.72 (14.62)	7.07 (0.23)	1.83 (1.02)	489.48 (120.42)	29.25 (3.37)	10.36 (0.36)	10.91 (2.27)	9.65 (0.32)	
VF - 3	1916.00 (174.43)	2.48 (0.31)	4.60 (0.09)	11.11 (2.16)	570.32 (122.28)	29.16 (2.93)	10.66 (0.30)	7.10 (0.15)	7.17 (0.64)	
VF - 4	2345.50 (223.44)	3.93 (1.12)	6.01 (0.10)	6.70 (2.12)	391.31 (128.06)	30.61 (3.06)	10.87 (0.31)	8.54 (0.17)	13.15 (0.90)	
VF - 5	2132.50 (95.43)	24.21 (2.74)	6.97 (0.04)	10.30 (2.72)	440.15 (103.38)	30.03 (2.75)	10.77 (0.24)	8.98 (0.23)	9.17 (0.24)	
VF - 6	1940.25 (294.89)	2.99 (0.52)	4.96 (0.12)	13.41 (4.01)	645.70 (189.52)	28.34 (3.14)	10.80 (0.38)	6.80 (0.15)	10.01 (1.27)	
VF - 7	972.50 (86.70)	5.39 (0.91)	6.81 (0.08)	0.13 (0.06)	206.37 (40.33)	25.58 (2.64)	9.79 (0.18)	5.32 (0.13)	5.76 (0.34)	
VF - 8	1431.00 (457.60)	7.37 (3.21)	6.13 (0.71)	2.42 (0.99)	352.54 (139.42)	28.00 (5.17)	10.24 (0.65)	6.24 (1.11)	7.99 (1.89)	
VF - 9	1006.67 (294.17)	8.90 (6.23)	6.08 (0.56)	0.71 (0.31)	738.85 (227.16)	25.87 (1.69)	9.74 (0.22)	5.15 (1.37)	6.89 (0.70)	
VF - 10	2365.00 (72.40)	7.24 (1.37)	6.46 (0.02)	10.01 (1.72)	853.61 (256.42)	30.31 (2.71)	10.89 (0.26)	9.04 (0.34)	14.66 (0.24)	
VF - 11	1821.75 (151.21)	3.63 (0.86)	6.00 (0.10)	11.08 (3.14)	629.30 (59.30)	28.81 (2.73)	10.58 (0.26)	7.33 (0.25)	9.95 (0.72)	
<b>VF Treatment</b>	<b>1780.22 (88.11)</b>	<b>8.96 (1.74)</b>	<b>6.11 (0.14)</b>	<b>6.74 (0.95)</b>	<b>517.21 (49.72)</b>	<b>28.62 (0.82)</b>	<b>10.49 (0.10)</b>	<b>7.64 (0.31)</b>	<b>9.50 (0.45)</b>	
R - 1	58.77 (7.36)	2.27 (0.18)	5.71 (0.05)	0.07 (0.07)	4.63*	1.62 (0.11)	1.12 (0.19)	1.62 (0.10)	2.57 (0.83)	
R - 2	81.25 (25.54)	2.88 (1.17)	4.89 (0.50)	0.04 (0.04)	19.63 (0.38)**	1.50 (0.26)	1.63 (0.46)	4.34 (2.54)	1.7 (0.14)	
R - 3	71.40 (1.98)	2.91(0.55)	5.64 (0.15)	0.06 (0.06)	8.41(1.48)	2.94 (0.89)	1.66 (0.33)	3.51(1.71)	2.67 (0.35)	
R - 4	44.75 (3.16)	1.76 (0.12)	5.72 (0.18)	0.00 (0.00)	6.01 (1.03)	1.18 (0.17)	1.38 (0.18)	1.71 (0.05)	1.77 (0.12)	
R - 5	58.23 (13.99)	2.07 (0.14)	5.78 (0.24)	0.00 (0.00)	5.96 (1.84)	1.34 (0.14)	1.50 (0.24)	1.64 (0.10)	3.44 (1.76)	
R - 6	42.75(1.76)	2.02 (0.29)	5.65 (0.17)	0.00 (0.00)	6.04 (1.16)	1.19 (0.20)	1.06 (0.13)	1.49 (0.05)	1.09 (0.08)	
R - 7	50.83 (8.58)	2.89 (0.11)	6.51 (1.11)	0.00 (0.00)	4.33 (0.87)	1.11 (0.17)	1.50 (0.21)	1.78 (0.10)	2.64 (1.31)	
R - 8	57.63 (5.86)	2.72 (0.62)	5.85 (0.31)	0.01 (0.01)	5.68 (1.57)**	1.55 (0.35)	1.42 (0.19)	1.74 (0.06)	3.02 (1.27)	
R - 9	73.18 (15.75)	2.46 (0.22)	5.42 (0.39)	0.00 (0.00)	7.54 (0.05)**	1.40 (0.29)	1.57 (0.11)	3.11 (1.36)	3.10 (2.00)	
R - 10	38.80 (5.91)	1.92 (0.12)	5.30 (0.09)	0.00 (0.00)	6.65 (2.33)	0.89 (0.23)	1.02 (0.10)	1.61 (0.08)	1.99 (0.60)	
R - 11	48.40 (3.87)	2.55 (0.33)	5.54 (0.06)	0.00 (0.00)	6.57 (2.29)	1.94 (0.45)	1.39 (0.18)	1.65 (0.08)	1.70 (0.13)	
R - 12	78.33 (9.47)	1.85 (0.30)	5.74 (0.61)	0.01 (0.01)	19.25 (0.92)**	1.84 (0.42)	1.93 (0.33)	3.23 (1.88)	1.54 (0.13)	
<b>R Treatment</b>	<b>58.41 (3.52)</b>	<b>2.35 (0.13)</b>	<b>5.64 (0.13)</b>	<b>0.01 (0.01)</b>	<b>7.67 (0.84)</b>	<b>1.51 (0.11)</b>	<b>1.43 (0.07)</b>	<b>2.27 (0.31)</b>	<b>2.25 (0.28)</b>	

**Figure legends.**

Fig. 1. Species/life-stage specific estimates of the occupancy probability for control (black) and mined (grey) sites. The points represent posterior means, the wide bands central 50% credible intervals, and the thin bands 95% central credible intervals.

Species/life stages denoted as *Gyrinophilus porphyriticus* (GP), adult *Desmognathus fuscus* (DF), adult *D. monticola* (DM), larval *Eurycea cirrigera* (EC-L), adult *E. cirrigera* (EC-A), larval *Desmognathus* spp. (DES-L) and *Pseudotriton ruber* (PR).

Fig. 2. Species/life-stage specific estimates of  $\Delta_{\gamma_S}$  (black) and  $\Delta_{\beta_S}$  (grey). The points represent posterior means, the wide bands central 50% credible intervals, and the thin bands 95% central credible intervals.

Species/life stages denoted as *Gyrinophilus porphyriticus* (GP), adult *Desmognathus fuscus* (DF), adult *D. monticola* (DM), larval *Eurycea cirrigera* (EC-L), adult *E. cirrigera* (EC-A), larval *Desmognathus* spp. (DES-L) and *Pseudotriton ruber* (PR).

Fig. 3. Species/life-stage specific estimates of overall abundance for control (black) and mined (grey) sites. The points represent posterior means, the wide bands central 50% credible intervals, and the thin bands central 95% credible intervals. Species/life stages denoted as *Gyrinophilus porphyriticus* (GP), adult *Desmognathus fuscus* (DF), adult *D. monticola* (DM), larval *Eurycea cirrigera* (EC-L), adult *E. cirrigera* (EC-A), larval *Desmognathus* spp. (DES-L) and *Pseudotriton ruber* (PR).

Fig. 4. Species/life-stage specific estimates of  $\Delta_{\gamma_S}$  (black) and  $\Delta_{\beta_S}$  (grey) computed in the sensitivity analysis. The points represent posterior means, the wide bands central 50% credible intervals, and the thin bands 95% central credible intervals. Species/life stages denoted as *Gyrinophilus porphyriticus* (GP), adult *Desmognathus fuscus* (DF), adult *D. monticola* (DM), larval *Eurycea cirrigera* (EC-L), adult *E. cirrigera* (EC-A), larval *Desmognathus* spp. (DES-L) and *Pseudotriton ruber* (PR).

Fig.1

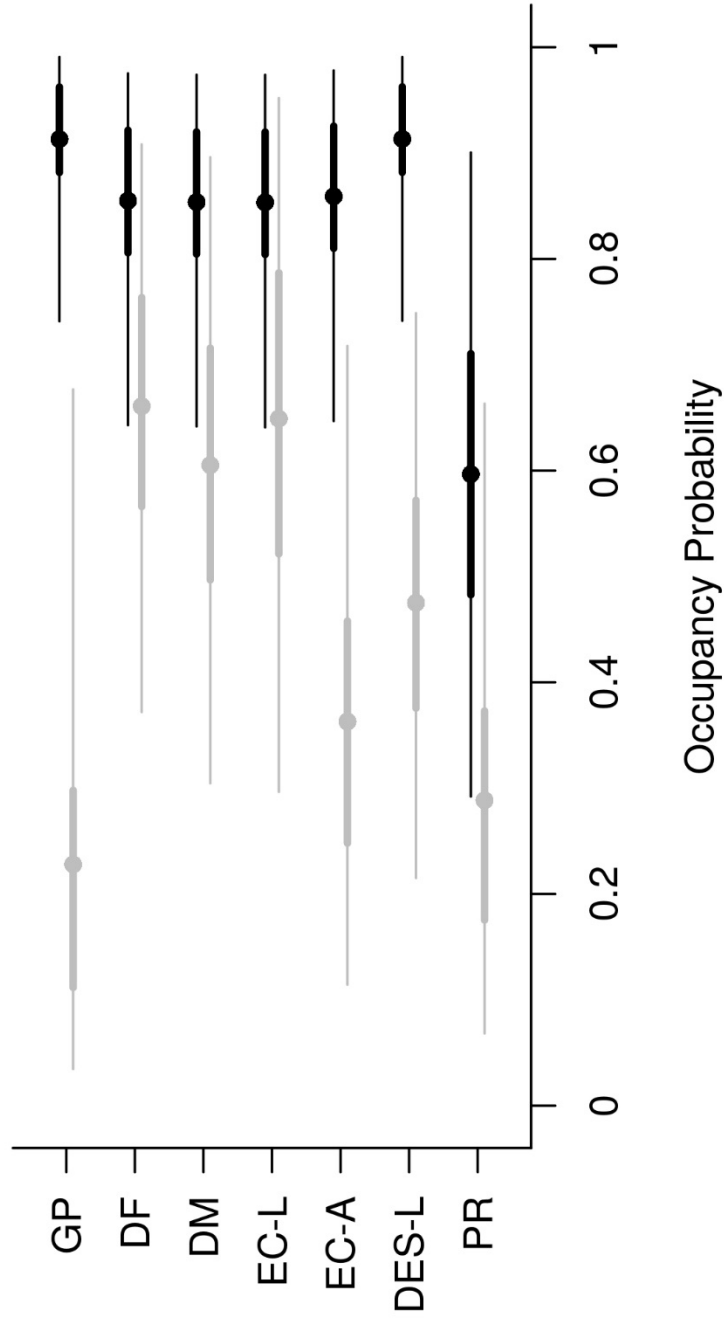




Fig. 2

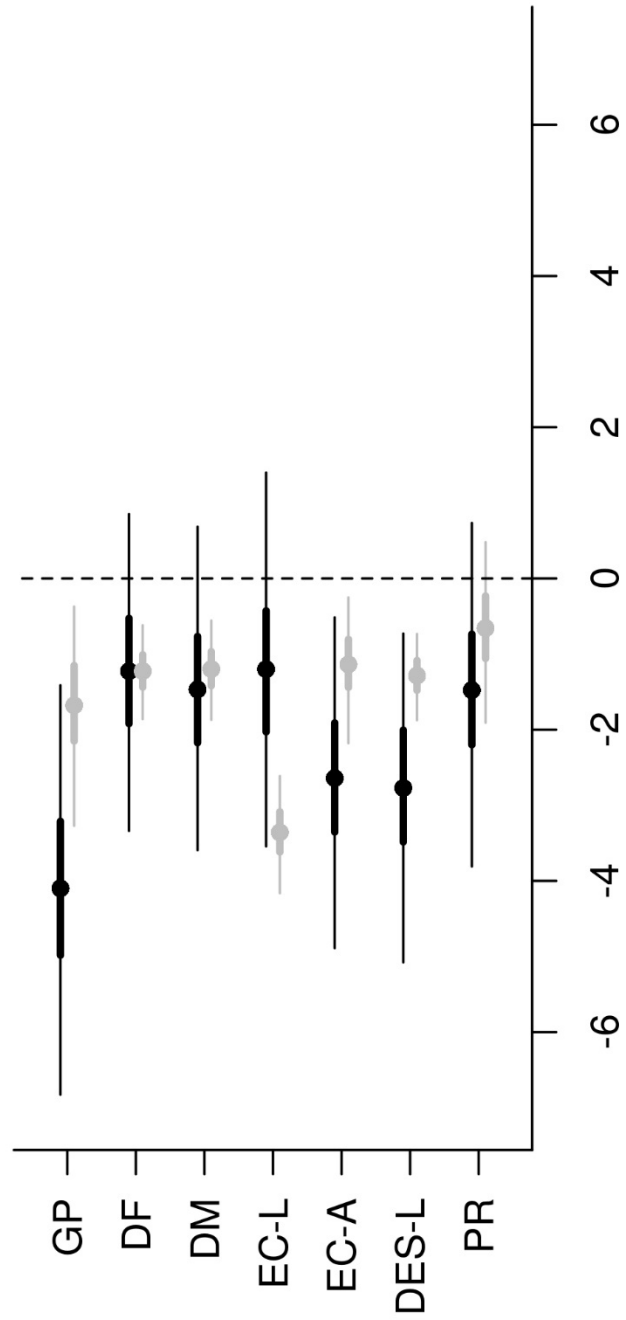


Fig. 3

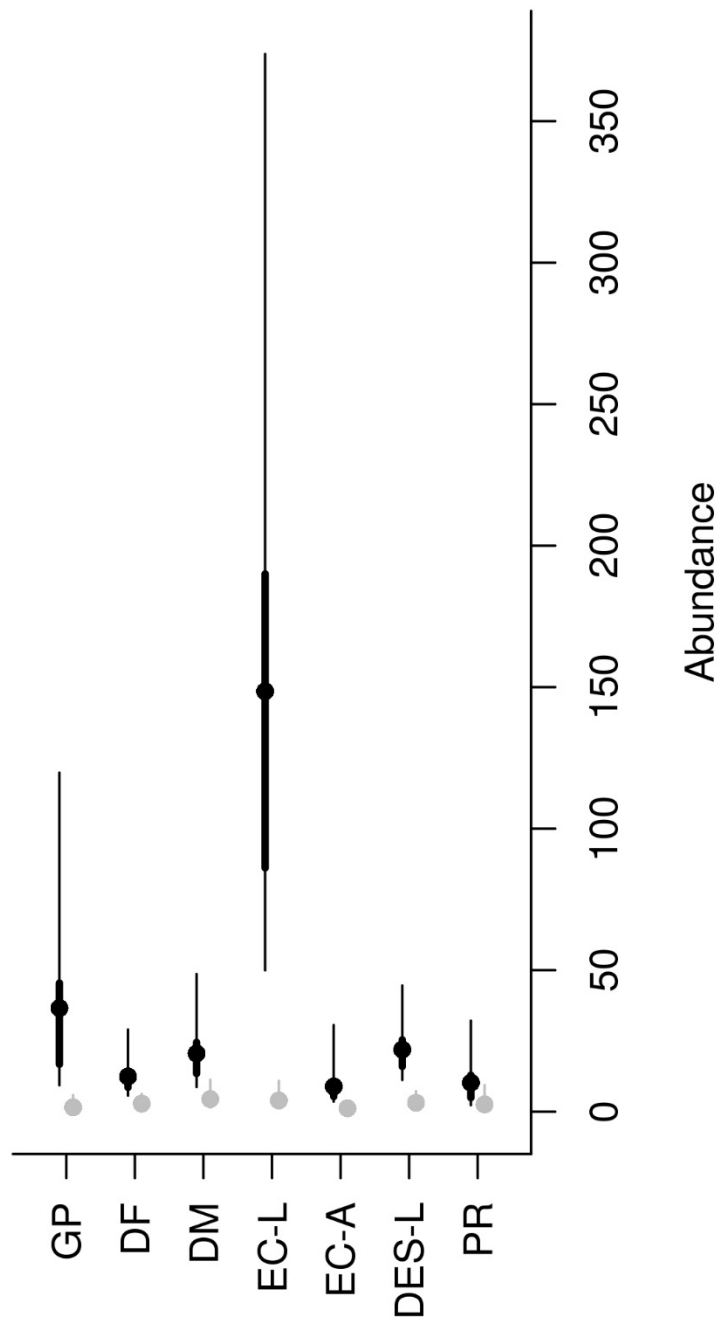
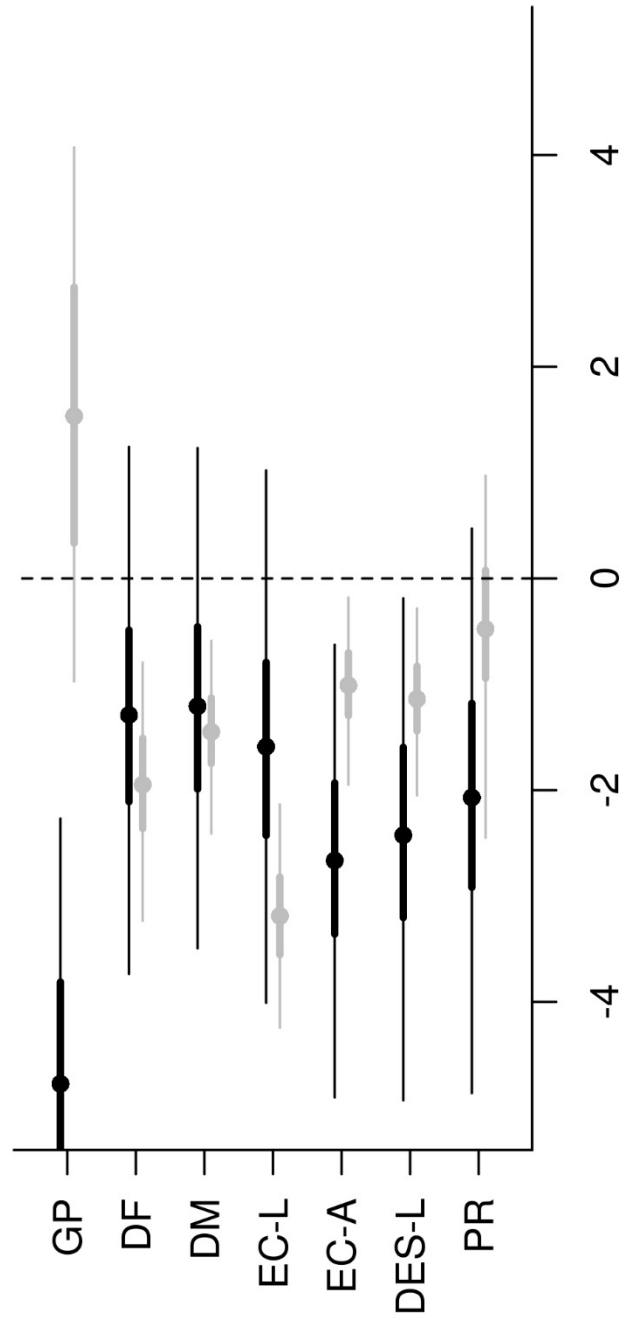
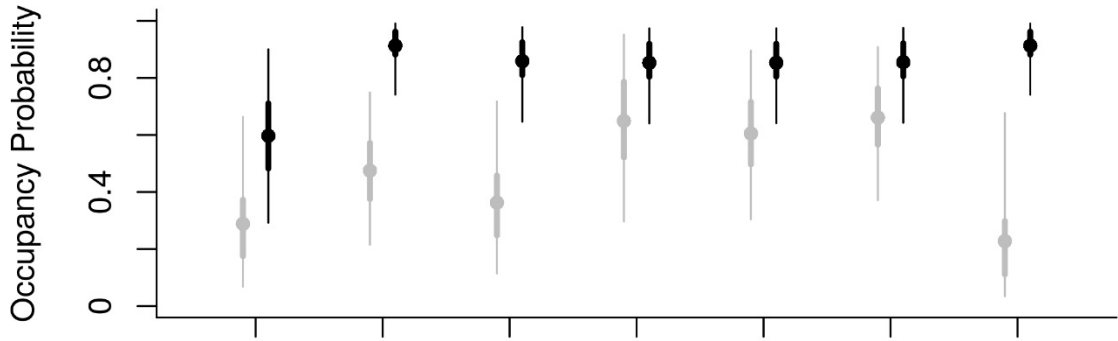


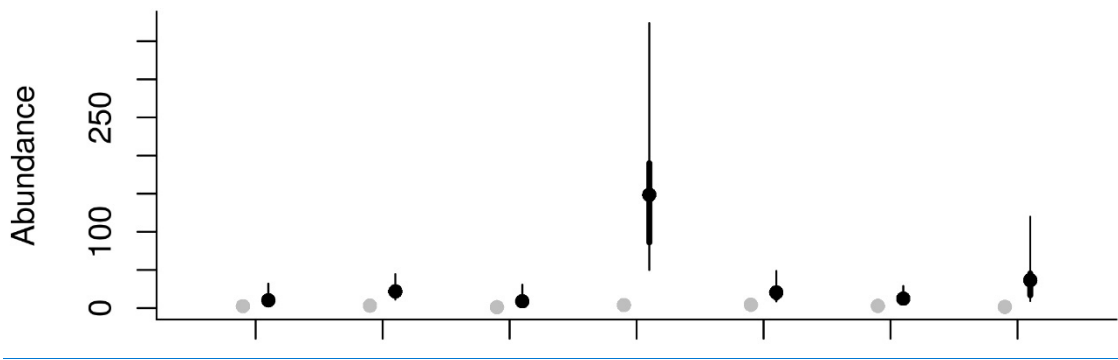
Fig 4



A



B



C

