

Effects of mountaintop removal mining and valley filling on the occupancy and abundance of stream salamanders

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Summary

1. Human-induced land-use changes are among the primary causes of ecosystem degradation and biodiversity loss. Across central Appalachia (USA), mountaintop removal mining and valley filling (MTR/VF) is the prevailing form of land-use change and represents a stressor to stream ecosystems. Salamanders are the dominant vertebrate in Appalachian headwater streams. Thus, we addressed the question: Is salamander occupancy and conditional abundance reduced in streams impacted by MTR/VF?

2. We conducted repeated counts of adult and larval salamanders within 10-m reaches in 11 valley-filled streams and 12 reference streams in south-eastern Kentucky. Relationships between occupancy, conditional abundance, and site type (MTR/VF vs. reference) were modelled using the hurdle model (Ecology, 94, 2013 and 1472), where occupancy is modelled separately from abundance while accounting for differences in per-individual detection probabilities among groups.

3. We found mean occupancy probabilities were >0.85 for all groups in reference reaches, whereas mean occupancy probabilities were relatively lower in MTR/VF reaches (ranging from 0.23 to 0.66). Posterior means of the difference in occupancy between site types were negative across all groups, although MTR/VF stream reaches were at least 95% less likely to be occupied by spring salamander *Gyrinophilus porphyriticus*, adult southern two-lined salamander *Eurycea cirrigera* and larval dusky salamanders *Desmognathus* compared to reference reaches.

4. Posterior means of the difference in conditional abundance between MTR/VF and reference stream reaches were negative across all groups; 95% credible interval for difference in conditional abundance covered zero for only one species (red salamander *Pseudotriton ruber*). After adjusting for goodness-of-fit, point estimates of differences in occupancy and conditional abundance still remained below zero for most species. Additionally, MTR/VF reaches had higher ion concentrations, total organic carbon and specific conductance compared to reference reaches.

5. *Synthesis and applications.* Our study concludes that mountaintop removal mining and valley filling (MTR/VF) reduces salamander occupancy and conditional abundance. Although the potential mechanisms responsible for reduction are numerous, our findings suggest a change in the current regulatory framework is needed to offset the impacts of MTR/VF on stream ecosystems and biota. Reclamation techniques that enhance conditions for vegetative succession within catchments may improve habitat on reclaimed surface mines.

Key-words: amphibians, Appalachia, coal mining, hurdle model, land use, mountaintop removal mining, reclamation, salamander, valley filling, water quality

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Introduction

Human-induced land-use changes are among the primary causes of ecosystem degradation, species population decline and losses in biodiversity (Pimm & Raven 2000). Land-use changes are often driven by the acquisition of natural resources, such as fibre, food and water (Foley *et al.* 2005). In many regions globally, the extraction of mineral resources via surface mining can also be an important driver of land-use change (i.e. Schmidt & Glaesser 1998; Bernhardt & Palmer 2011; Kujala *et al.* 2015). Thus, understanding the environmental and ecological costs of surface mining has become an issue of global significance (Bridge 2004).

Mountaintop removal mining, a form of surface mining that involves the removal of significant amounts of rock to access shallow coal seams, is a well-known stressor to ecosystems throughout the Central Appalachia region of the eastern USA (Bernhardt & Palmer 2011; Wickham *et al.* 2013). Aquatic ecosystems are particularly impacted as mountaintop removal mining often results in the complete or partial burial of low-order streams via valley filling, the process by which rock (i.e. overburden materials) is discarded from the mine site into adjacent valleys (Bernhardt & Palmer 2011). Valley filling alters stream water chemistry; leaching and surface run-off from the unweathered, overburden materials leads to increased specific conductance and ion concentrations, elevated levels of total dissolved solids and altered pH compared to reference streams (Palmer *et al.* 2010; Griffith *et al.* 2012). Changes to stream catchments from mining, including forest removal and soil compaction, result in larger storm run-off coefficients, greater total run-off, higher peak hourly run-off rates, and increases in sedimentation compared to reference streams (Negley & Eshleman 2006; Simmons *et al.* 2008; Ferrari *et al.* 2009). Collectively, mountaintop removal mining and valley filling (MTR/VF) results in long-lasting degradation to headwater streams and their catchments (Simmons *et al.* 2008; Bernhardt & Palmer 2011; Wickham *et al.* 2013). Although regulatory mechanisms, including the U.S. Clean Water Act (CWA) and the Surface Mining Control and Reclamation Act (SMCRA), are in place to minimize impacts, the US Environmental Protection Agency (EPA; US EPA 2011) estimated that SMCRA permits from 1992 and 2002 authorized the destruction of ~1900 km of headwater streams in Central Appalachia.

Mountaintop removal mining is widespread in the central Appalachian states of Kentucky, Tennessee, Virginia and West Virginia, a region known for high levels of stream biodiversity (Bernhardt & Palmer 2011). Salamanders represent the dominant vertebrates in low-order streams in Appalachia where they can reach exceptional population densities (i.e. Peterman, Crawford & Semlitsch 2008) and drive numerous ecosystem-level processes. For example, salamanders convert large quantities of

invertebrate prey to vertebrate biomass; they serve as prey for many animals and influence nutrient cycling and energy flow within streams and from aquatic to terrestrial environments (Davic & Welsh 2004; Keitzer & Goforth 2013; Milanovich, Maerz & Rosemond 2015). It is well documented that salamander populations are sensitive to land-use changes that impact streams and their catchments (e.g. Crawford & Semlitsch 2007; Barrett & Price 2014). Despite the ubiquity of MTR/VF in central Appalachia, only Wood & Williams (2013a) examined the effects of MTR/VF on relative abundances (i.e. total number of individuals counted across all species) of adults and larva; they found relative abundances were greater in three reference streams compared to three streams impacted by MTR/VF. However, counts of some species (i.e. northern dusky salamander *Desmognathus fuscus* (Rafinesque, 1820)) were similar between MTR/VF streams and reference streams (Wood & Williams 2013a). Thus, investigations that not only focus on species- and stage-specific responses but also examine the effects in different locations across the central Appalachian ecoregion may be necessary to fully ascertain impacts of MTR/VF on salamander populations and aquatic ecosystems.

In this study, we evaluated the effects of MTR/VF on stream salamanders in the eastern Kentucky coalfields. Specifically, we modelled counts of adult and larva of five salamander species using binomial mixture models assuming a zero-inflated Poisson distribution of abundance at each stream. The zero-inflated Poisson model allows for occupancy of the site to be modelled separately from the abundance, given the site was occupied, while accounting for variation in per-individual detection probability (e.g. Dorazio, Martin & Edwards 2013). We ask the basic question: Is salamander occupancy and abundance given occupancy (hereafter conditional abundance) reduced in stream reaches impacted by MTR/VF compared to reference stream reaches? In addition, we report water chemistry and other physical characteristics of valley-filled streams and reference streams to determine stressors potentially responsible for patterns of species' occupancy and conditional abundance.

Materials and methods

STUDY SITES

Our study sites consisted of 23 headwater streams located in the interior rugged section of the Cumberland Plateau in Breathitt and Knott counties, Kentucky USA (Fig. 1). Specifically, we sampled stream salamanders at 11 streams, partially buried by overburden materials (i.e. valley-filled), located on the reclaimed Laurel Fork surface mine (4144091.438 N 307635.435 E Zone 17) and 12 reference streams in approximately 80-year-old, second-growth forest on the University of Kentucky's Robinson Forest (see Appendix S1, in Supporting Information for photographs of streams). Robinson Forest is located directly north-east of the Laurel Fork surface mine; Laurel Fork surface mine was part of

Robinson Forest until the early 1990s when mineral rights of the property were sold. Thus, the valley-filled streams and reference streams shared many attributes (i.e. similar vegetation composition and age, elevation, subsurface geology) prior to surface mining (Wiken, Jiménez Nava & Griffith 2011). Laurel Fork was surface mined from the late 1990s to the early 2000s, and released from bond in November 2007 signifying reclamation was satisfied.

Dominant vegetation within stream catchments on the Laurel Fork surface mine consisted of nitrogen-fixing herb sericea lespedeza *Lespedeza cuneata* (Don, 1832) and grasses (e.g. tall fescue *Schedonorus arundinaceus* (Roemer & Schultes, 1817)), with autumn olive *Elaeagnus umbellata* (Thunberg, 1784), Virginia pine *Pinus virginiana* (Mill., 1768), white oak *Quercus alba* (L., 1753) and black locust *Robinia pseudoacacia* (L., 1753) scattered throughout the landscape (see Fritz *et al.* 2010 for additional details). Although average percentage of forest cover within stream catchments was low at study sites on the Laurel Fork surface mine (25%), riparian zones and adjacent terrestrial habitats were primarily forested. Reference streams were located on the main block of Robinson Forest and were dominated by second-growth, mixed, mesophytic forests (i.e. tulip tree *Q. alba*, *Lirio-*

dendron tulipifera (L., 1753), eastern hemlock *Tsuga canadensis* (Carrière, 1855)). Average forest cover within reference stream catchments exceeded 99%. The average catchment sizes were similar between stream types (reference stream = 24.70 ha; MTR/VF stream = 24.51 ha). See Muncy *et al.* (2014) for additional details about study sites.

DATA COLLECTION

We conducted repeated counts of salamanders within 10-m stream reaches in all MTR/VF and reference streams. Reaches were chosen based on their similarity of width, depth and water flow and all reaches included a pool, run and riffle section. Selecting sampling sites in this manner ensured in-stream habitat would be appropriate for salamanders. Most MTR/VF stream reaches were located at the base of the valley fill. Despite the similarities in mean width (reference streams = 130.6 cm; MTR/VF streams = 122.6 cm) and mean depth (reference streams = 6.76 cm; MTR/VF streams = 7.45 cm) between reference and MTR/VF reaches, the average number of cover objects (rocks > 50 mm in diameter and logs) within each reach were greater at reference streams (48 cover objects) than MTR/VF streams (24 cover objects; see Muncy *et al.* 2014).

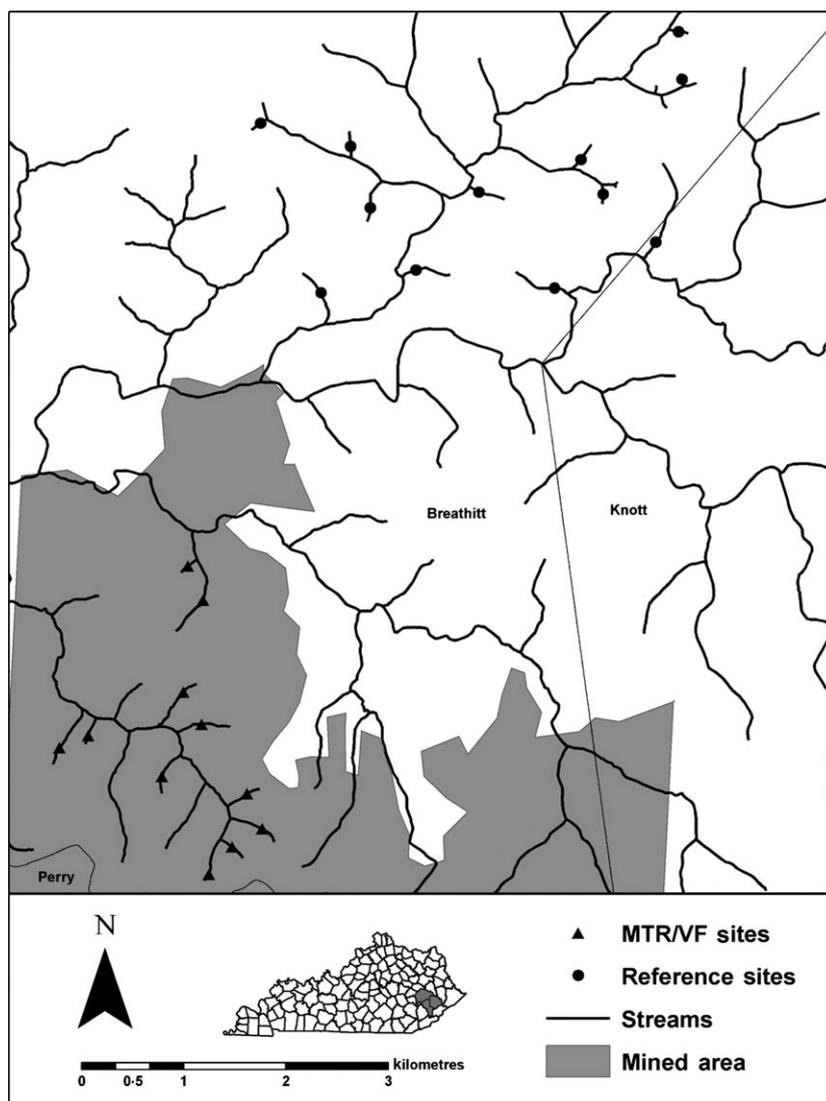


Fig. 1. Location of stream reaches sampled for salamanders in south-eastern Kentucky, USA. Eleven streams – mountaintop removal mining and valley filling (MTR/VF) (denoted with a triangle) were located on the reclaimed Laurel Fork surface mine, whereas 12 sites (denoted with a circle) were located in the University of Kentucky's Robinson Forest.

We collected a 50 mL water sample from each stream reach prior to each sampling event to evaluate the effects of MTR/VF on stream water chemistry. In the laboratory, the water samples were analysed for concentrations of calcium (Ca), magnesium (Mg), potassium (K), sodium (Na), sulphates (SO_4^{2-}), total organic carbon (TOC), pH and specific conductance (μS); sampling and analyses were performed in accordance with standard methods (Greenberg, Clesceri & Eaton 1992). These data have been formally analysed in Muncy *et al.* (2014), although we present site-specific averages and standard errors in Table 1.

Salamander count data were obtained through area-constrained active searches, specifically dipnetting and stream bank searches within the 10-m reach (see Price, Browne & Dorcas 2012). Dipnetting consisted of one sampler, moving from downstream to upstream, actively netting for adult and larval salamanders around and under cover objects (i.e. logs, rocks) and leaf litter within the stream. Stream bank searches also were conducted by one person; these searches included close examination of cover objects and other material within 1-m of the wetted width of the stream reach. Active searches were conducted during daylight hours and in baseflow conditions. We limited dipnetting and bank searches to 30 and 15 min, respectively. After searches were complete, all salamanders were identified to species, life stage (larva or adult), counted and released back into the 10-m reach. We sampled each 10-m reach four times (i.e. usually monthly; mean number of days between samples = 21.69) from March through June 2013.

We also recorded several sampling covariates at each stream reach prior to each active search to account for the conditions that may influence our ability to detect salamanders. We assumed that per-individual detection rate (i.e. probability of detecting an individual that is present in a reach and available for detection) of salamanders may be influenced by four covariates: number of days since last precipitation event (i.e. 0, 1, 2, 3 days), water temperature ($^{\circ}\text{C}$), date (i.e. Julian day of year) and number of cover objects (i.e. rocks > 50 mm diameter and woody debris > 80 mm in diameter) within the sampling reach. Water temperature, number of days since last precipitation and number of cover objects have been previously noted as important predictors of detection of stream salamanders (Johnson & Goldberg 1975; Orser & Shure 1975; Price, Browne & Dorcas 2012). We recorded date because activity, and potentially detection, may change over the 4 months of sampling.

STATISTICAL ANALYSIS

To examine the effects of mountaintop removal mining on stream salamander occupancy and conditional abundance, we modelled counts of salamanders using the extended binomial mixture model of Dorazio, Martin & Edwards (2013). This model assumes an underlying zero-inflated Poisson distribution of the population size at each stream which Dorazio, Martin & Edwards (2013) call the hurdle model. The hurdle model is parameterized so that occupancy of the streams is modelled separately from the abundance among occupied locations (i.e. conditional abundance), while still accounting for differences in per-individual detection probabilities among groups. Assumptions of this model are that the population at each stream is closed to entry (i.e. birth, immigration) and exit (i.e. death, emigration), equal catchability among salamanders and independence among study sites. This model was appropriate for

our salamander repeated count data because it accounts for the excess of zeros in observed counts in the MTR/VF stream reaches; failure to account for this variation can substantially bias abundance estimates (see Dorazio, Martin & Edwards 2013).

To conduct our analysis, we first separated salamander count data by species and in some instances stage (i.e. adult vs. larva). Specifically, we separately analysed counts for only adult *D. fuscus*, seal salamander *D. monticola* (Dunn, 1916) and southern two-lined salamander *E. cirrigera* (Green, 1830). We combined counts of adult and larval forms of spring salamander *G. porphyriticus* (Green, 1827) as well as counts of red salamander *P. ruber* (Latreille, 1801), as all but one of the captures of these two species were larvae. Finally, we analysed larval counts of *E. cirrigera* and combined counts for larval *D. fuscus* and *D. monticola* due to difficulty of separating *Desmognathus* species in larval form; collectively we refer to these as *Desmognathus* larvae. In total, we had seven separate groups of salamanders.

We first modelled occupancy of group s at site i , O_{si} , as a Bernoulli random variable (e.g. MacKenzie *et al.* 2003) where the probability that site i was occupied by group s , ψ_{si} , was allowed to vary between MTR/VF ($m_i = 1$) and reference stream reaches ($m_i = 0$) modelled on the logistic scale, given by:

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

The difference in ψ_{si} between MTR/VF and reference stream reaches for group s was measured through a pairwise contrast such that $\Delta\gamma_s = \gamma_{s1} - \gamma_{s2}$. Differences in occupancy between MTR/VF and reference streams were assessed by considering whether or not the central 50% and 95% credible intervals for $\Delta\gamma_s$ covered zero.

Abundance for each group s at site i (N_{si}) was then modelled conditional on occupancy. If the site was unoccupied by a given group s , we considered the abundance to be zero. Otherwise, the abundance at site i was modelled as a zero-truncated Poisson random variable with rate parameter λ_{si} such that:

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

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

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

The distribution of conditional abundance was allowed to vary between reference and MTR/VF streams and among groups so that:

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

The difference in conditional abundance between MTR/VF and reference streams for group s was measured as $\Delta\beta_s = \beta_{s1} - \beta_{s2}$; differences were assessed by considering whether or not the 50% and 95% credible intervals for $\Delta\beta_s$ covered zero.

Accounting for both occupancy and conditional abundance, the distribution for the overall abundance of group s at site i was given by the probabilities:

Table 1. Site-specific average (\pm SE) for select water chemistry attributes, specific conductance, total organic carbon (TOC), pH, manganese (Mn), sulphate (SO_4), calcium (Ca), magnesium (Mg), potassium (K) and sodium (Na), at mountaintop removal and valley-filled (MTR/VF) and reference streams (R) located in eastern Kentucky, USA. Overall treatment means are represented by mountaintop removal mining and valley filling (MTR/VF) or reference (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^{-1} were below detection limits

| Study Site | Specific conductance ($\mu\text{S cm}^{-1}$) | TOC (mg L^{-1} C) | pH [H+] | Mn (mg L^{-1}) | SO_4 (mg L^{-1}) | Ca (mg L^{-1}) | Mg (mg L^{-1}) | K (mg L^{-1}) | Na (mg L^{-1}) |
|------------------|--|-----------------------------|-------------|---------------------------|--------------------------------------|---------------------------|---------------------------|--------------------------|---------------------------|
| MTR/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) |
| MTR/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) |
| MTR/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) |
| MTR/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) |
| MTR/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) |
| MTR/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) |
| MTR/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) |
| MTR/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) |
| MTR/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) |
| MTR/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) |
| MTR/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) |
| MTR/VF Treatment | 1780.22 (88.11) | 8.96 (1.74) | 6.11 (0.14) | 6.74 (0.95) | 517.21 (49.72) | 28.62 (0.82) | 10.49 (0.10) | 7.64 (0.31) | 9.50 (0.45) |
| 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) |
| R Treatment | 58.41 (3.52) | 2.35 (0.13) | 5.64 (0.13) | 0.01 (0.01) | 7.67 (0.84) | 1.51 (0.11) | 1.43 (0.07) | 2.27 (0.31) | 2.25 (0.28) |

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

This represents a zero-inflated Poisson distribution in which occupancy may be less than expected under a strict Poisson model for abundance. The expected abundance of group s at site i (i.e. $E(N_{si})$) combining the models for both occupancy and conditional abundance was:

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

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

Salamander counts on the four visits were assumed to follow independent binomial distributions conditional on the site-specific population size. Detection probabilities were allowed to depend on environmental covariates but were otherwise assumed to be the same for MTR/VF and reference streams. Individual salamanders were assumed to behave independently within and between sampling occasions, so the number of captures of species/life stage s at site j on occasion i followed a binomial distribution:

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

The probability of capture was modelled on the logit scale as:

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

where x_{sij} was a vector of sampling covariates and α_s the associated vector of coefficients for species/life stage s . We assumed that per-individual detection rate of salamanders may be influenced by four covariates: water temperature ($^{\circ}\text{C}$), number of days since last precipitation event, Julian date and number of cover objects within the sampled reach. All sampling covariates were standardized so the mean of the population was 0 and the standard deviation was 1.

Dorazio, Martin & Edwards (2013) provided maximum likelihood methods to fit the hurdle models. However, we chose to model each group separately in a Bayesian framework using Markov chain Monte Carlo (MCMC) sampling in OpenBUGS to generate samples from the posterior distribution (Lunn *et al.* 2009; see Appendix S2). All parameters were assigned non-informative prior distributions. Three parallel chains were run in OpenBUGS for each model so that convergence could be assessed via the Gelman–Rubin diagnostic (Gelman & Rubin 1992). Each chain was run for 50 000 iterations in total, the first 25 000 were removed as burn-in, and the final 25 000 were thinned by a factor of 5. This provided a total of 15 000 samples from which we approximated posterior summary statistics including the mean, standard deviation, and 50% and 95% credible intervals for each of the model parameters. Posterior summary statistics were also computed for derived quantities including the occupancy probability and abundance of each group at each of the sites.

Goodness-of-fit was conducted after the initial modelling stage by computing Bayesian P -values based on the contributions to the completed data likelihood function for each species or life stage and each site. The complete data likelihood function was

constructed by treating occupancy and abundance as if they were part of the observed data. In particular, we constructed three discrepancy measures from the log of the likelihood components modelling the abundance (including occupancy) and the observed values conditional on abundance, and from the overall likelihood (i.e. see Kéry & Schaub 2012). Bayesian P -values were then computed by comparing the discrepancy measures obtained for both the observed data and simulated data generated on each iteration of the MCMC algorithm (see Gelman, Meng & Stern 1996). We considered Bayesian P -values ≤ 0.10 for any of the three discrepancy measures as indications of potential lack-of-fit. For each group, the analysis (see occupancy and abundance analyses above) was repeated after removing all sites producing a Bayesian P -value ≤ 0.10 for any of the three discrepancy measures (see Appendix S3).

Results

We counted 92 *D. fuscus* adults, 117 *D. monticola* adults, 46 *E. cirrigera* adults, 106 *G. porphyriticus*, 22 *P. ruber*, 190 *E. cirrigera* larvae and 188 *Desmognathus* larvae in reference stream reaches. Salamander counts in MTR/VF reaches resulted in 25 adult *D. fuscus*, 19 adult *D. monticola*, 4 adult *E. cirrigera*, 2 *G. porphyriticus* larvae, 5 *P. ruber* larvae, 11 *E. cirrigera* larvae and 25 *Desmognathus* larvae. We found differences in our group estimates of occupancy between reference sites and MTR/VF reaches. With the exception of *P. ruber*, estimated occupancy probabilities were >0.85 for all species and stages in reference reaches (Fig. 2a). Estimated occupancy probabilities were relatively low in MTR/VF reaches, ranging from 0.23 (95% credible interval (CI) = 0.03–0.68) for *G. porphyriticus* to 0.66 (95% CI = 0.37–0.91) for *D. fuscus* (Fig. 2a). Despite the variation, posterior means of $\Delta\gamma_s$ for all groups were negative, suggesting decreased occupancy in reaches impacted by MTR/VF (Fig. 2c). Upper bounds of the 95% credible intervals for $\Delta\gamma_s$ were below zero for three species, *G. porphyriticus*, adult *E. cirrigera* and larval *Desmognathus*. Although the upper bounds of the 50% credible intervals for *D. fuscus*, *D. monticola*, larval *E. cirrigera* and *P. ruber* were well below zero, we could not draw strong conclusions about differences in occupancy for these species because the 95% credible intervals covered zero.

Our estimates of conditional abundance also exhibited both group and site type differences (Fig. 2b). In reference reaches, estimated mean conditional abundance ranged from 148.53 (95% CI = 49.95–373.70) larval *E. cirrigera* to 8.92 (95% CI = 3.56–30.65) adult *E. cirrigera* per 10-m reach (Fig. 2b). Conversely, across all groups, mean conditional abundance was lower in MTR/VF reaches (Fig. 2b). The 95% credible interval for $\Delta\beta_s$ covered zero for only one species (*P. ruber*) and posterior means $\Delta\beta_s$ for all species and stages were negative, suggesting reduced conditional abundance in reaches impacted by MTR/VF (Fig. 2c).

Sampling covariates influenced per-individual detection probabilities for some groups. *Gyrinophilus porphyriticus*

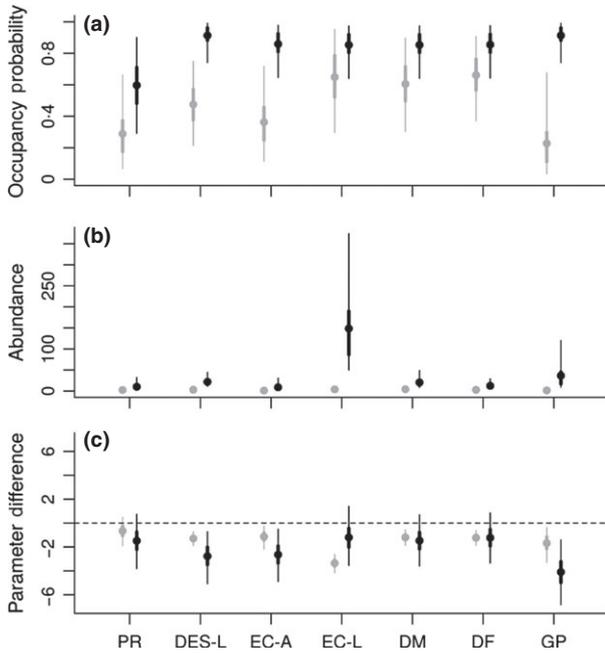


Fig. 2. Salamander group estimates of (a) occupancy probability and (b) conditional abundance for reference (black) and mined (grey) sites in south-eastern Kentucky, USA. (c) The differences in occupancy ($\Delta\gamma_s$, black) and conditional abundance ($\Delta\beta_s$, grey), between mountaintop removal mining and valley filling, and reference streams. The points represent posterior means, the wide bands central 50% credible intervals, and the thin bands 95% central credible intervals. Groups are denoted as *Pseudotriton ruber* (PR), larval *Desmognathus* spp. (DES-L), larval *Eurycea cirrigera* (EC-L), adult *E. cirrigera* (EC-A), adult *D. monticola* (DM), adult *Desmognathus fuscus* (DF), and *Gyrinophilus porphyriticus* (GP). We combined counts of adult and larval forms of GP and PR as all but one of the captures of these two species were larvae.

individuals were detected with greater frequency as date increased ($\alpha = 0.31$, 95% CI = 0.12–0.54 (parameter estimates are from standardized covariates)). Adult *D. fuscus* individuals were detected less frequently as the number of in-stream cover objects increased ($\alpha = -0.53$ (95% CI = -0.87 to -0.19)). Per-individual detection of larval *E. cirrigera* increased throughout the sampling period ($\alpha = 0.70$ (95% CI = 0.53–0.88)) and decreased as the number of in-stream cover objects increased ($\alpha = -0.77$ (95% CI = -0.96 to -0.58)). Adult *E. cirrigera* individuals were detected less frequently as date increased ($\alpha = -1.98$ (95% CI = -2.57 to -1.19)) and more frequently as water temperature increased ($\alpha = 0.71$ (95% CI = 0.08–1.41)).

The goodness-of-fit assessment for all groups indicated lack-of-fit of the models at one or more of the study reaches. For each group, the analysis (see occupancy and abundance analysis) was repeated after removing all reaches producing a Bayesian *P*-value ≤ 0.10 for any of the three discrepancy measures (i.e. occupancy (O_{si}), conditional abundance (N_{si}) and the observed counts on each occasion given abundance (Y_{si})). In all but one case, the point estimates of $\Delta\gamma_s$ and $\Delta\beta_s$ remained below zero, though the estimated differences were closer to zero than

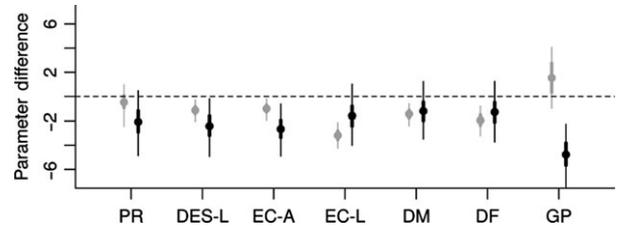


Fig. 3. The differences in occupancy ($\Delta\gamma_s$, black) and conditional abundance ($\Delta\beta_s$, grey), between mountaintop removal mining and valley-filled, and reference streams computed in the sensitivity analysis. The points represent posterior means, the wide bands central 50% credible intervals, and the thin bands 95% central credible intervals. Groups are denoted as *Pseudotriton ruber* (PR), larval *Desmognathus* spp. (DES-L), larval *Eurycea cirrigera* (EC-L), adult *E. cirrigera* (EC-A), adult *D. monticola* (DM), adult *Desmognathus fuscus* (DF), and *Gyrinophilus porphyriticus* (GP). We combined counts of adult and larval forms of GP and PR as all but one of the captures of these two species were larvae.

in the initial analysis and credible intervals were wider, which was expected given the removal of study reaches (Fig. 3; see Fig. S2). The single exception was the point estimate of the difference in conditional abundance for *G. porphyriticus* which was above zero.

Reference reaches had consistently lower average concentrations of calcium (Ca), magnesium (Mg), potassium (K), sodium (Na) and sulphates (SO_4^{2-}) than those recorded on MTR/VF sites (Table 1). Additionally, total organic carbon (TOC) and pH tended to be elevated in MTR/VF reaches (Table 1). In particular, site-specific averages of specific conductance ranged from 972.50 (± 86.70) to 2365.00 (± 72.40) $\mu\text{S cm}^{-1}$ in MTR/VF reaches whereas, in reference reaches, site-specific averages ranged from 38.80 (± 5.91) to 81.25 (± 25.54) $\mu\text{S cm}^{-1}$ (Table 1).

Discussion

Stream salamander populations are sensitive to a variety of land-use changes (Connette & Semlitsch 2013; Barrett & Price 2014; Surasinghe & Baldwin 2015); however, few studies have examined their response to MTR/VF, the primary type of land-use disturbance in Central Appalachia (but see Wood & Williams 2013a; Muncy *et al.* 2014). We found *G. porphyriticus*, adult *E. cirrigera* and larval *Desmognathus* were unequivocally less likely to occupy streams impacted by MTR/VF, and, with the exception of *P. ruber*, all other species and life stages had clearly reduced conditional abundance in MTR/VF stream reaches. Results from the sensitivity analysis were qualitatively the same as those obtained from the original analysis for most groups (i.e. *D. fuscus* and *D. monticola* adults, *E. cirrigera* larva, and *P. ruber*). However, results for a few species, particularly *G. porphyriticus* and *Desmognathus* larva, showed more sensitivity to the sites with potential lack-of-fit. Yet, these results were not surprising given that *G. porphyriticus* was detected at only one

MTR/VF reach which was removed during the sensitivity analysis. Nine of the 23 reaches were excluded from the sensitivity analysis for *Desmognathus* larvae, greatly reducing the amount of data and suggesting either a violation of the closure assumption or that the detection model for *Desmognathus* larvae was not appropriate. Nonetheless, our robust modelling approach provided statistically unbiased information indicating a general intolerance of stream salamanders to land-use disturbance created by MTR/VF operations at our study location.

This reduced occupancy and conditional abundance of salamanders in streams impacted by MTR/VF is likely due to the multitude of stressors in both streams and stream catchments. First, stream catchments on mined land often have little forest cover due to poor soil conditions following mining and/or restoration techniques (Chaney, Pope & Byrnes 1995). Indeed, our MTR/VF stream catchments contained, on average, only 25% forest cover with the remaining land use in grasses and shrubs (see study sites above). Although revegetation of mined land is required under SMCRA, this process typically involves hydroseeding the recontoured land with grasses (Zipper *et al.* 2011). Succession from grassland to pre-mining forest communities has generally not been successful (Zipper *et al.* 2011); this results in a reduction in leaf litter and woody debris cover on reclaimed MTR sites (Wood & Williams 2013b). Additionally, remaining forest patches are often highly fragmented, with high edge to interior ratios, creating drier, warmer and windier conditions than interior forests (Harper *et al.* 2005; Wickham *et al.* 2013). Because salamanders are prone to desiccation, humidity and temperature, adequate moisture and cover strongly influence the spatial distribution and local abundances in terrestrial environments (Peterman & Semlitsch 2013). Wood & Williams (2013b) suggested that decreased microhabitat availability and warmer and drier microclimatic conditions contributed to low abundances of terrestrial salamanders on reclaimed MTR mines. It is likely that poor conditions contributed to lower occupancy rates and conditional abundance of some stream-associated species as some species (i.e. *E. cirrigera*, *P. ruber*) extensively inhabit terrestrial habitats adjacent to streams during the nonbreeding season whereas other species are known to forage and disperse through terrestrial environments. More broadly, consequences of degraded terrestrial habitat conditions on reclaimed mined land include a reduction in ecological connectivity and disruption of aquatic–terrestrial trophic linkages (Greene, Lowe & Likens 2008; Grant *et al.* 2010).

It is well documented that land use within stream catchments strongly influences in-stream processes and habitat conditions (Allan 2004). SMCRA protocols advocate for the compaction of overburden materials or remaining topsoil in terrestrial environments to reduce the probability of landslides (Acton *et al.* 2011). As a result of soil compaction (and low evapotranspiration), MTR catchments often have reduced infiltration rates, which lead to

increased storm run-off, higher baseflow volumes and greater potential for flooding (Negley & Eshleman 2006; Ferrari *et al.* 2009; Fox 2009). High stream flows and flooding have been linked to reduced survival of larval *E. cirrigera* (Barrett *et al.* 2010) and reduced recruitment in *G. porphyriticus* (Lowe 2012). Additionally, studies have found a higher proportion of fine sediments and embedded substrate in streams impacted by MTR/VF (Fox 2009). Siltation and reduction in habitat complexity have been connected to reduced abundances of various salamander species, especially larvae (Redmond 1980; Lowe, Nislow & Bolger 2004; Wood & Williams 2013a). The reduced abundance of in-stream salamander populations may have significant ecological consequences to stream ecosystems as larvae regulate invertebrate detritivore populations and indirectly slow the release of nutrients to downstream areas (Keitzer & Goforth 2013; Milanovich, Maerz & Rosemond 2015).

We found consistently greater concentrations of ions and elevated specific conductance in MTR/VF reaches compared to reference reaches, which is likely due to the weathering of the overburden material. Elevated specific conductance (i.e. $>500 \mu\text{S cm}^{-1}$) within streams has been positively correlated with decreased macroinvertebrate abundance (Pond *et al.* 2008). Macroinvertebrates represent important prey items to salamanders (Keitzer & Goforth 2013) and may be a possible explanation for decreased occupancy and conditional abundance in streams impacted by MTR/VF. Increased levels of specific conductance have been shown to influence amphibian behaviour (Karraker, Gibbs & Vonesh 2008), corticosterone levels and feeding (Chambers 2011). Embryonic and larval survival of amphibians has been found to be reduced at moderate (500 μS) and high specific conductivities (3000 μS) (Karraker, Gibbs & Vonesh 2008), although we are not aware of studies conducted on plethodontid salamanders. Future studies are needed to disentangle the aforementioned stressors potentially responsible for low conditional abundance and occupancy in MTR/VF reaches.

MANAGEMENT RECOMMENDATIONS

Based on our results and those from other studies (i.e. Wood & Williams 2013a), it appears stream salamander populations, along with macroinvertebrates (Pond *et al.* 2008) and fishes (Hopkins & Roush 2013; Hitt & Chambers 2014), have reduced levels of occupancy and abundance in streams impacted by MTR/VF. Palmer *et al.* (2010) note that full recovery of stream biota post-restoration has not been documented. Collectively, this suggests that the protection of headwater streams governed by the CWA (section 404) and the reclamation and mitigation of terrestrial and aquatic ecosystems required under SMCRA have been unsuccessful in preventing habitat loss, circumventing population declines and promoting recovery of aquatic ecosystems and stream biota. Thus, new or

enhanced regulations governing MTR/VF are needed. In addition to changes in regulations, the utilization of alternative reclamation techniques could offer management strategies beneficial to aquatic ecosystems and semi-aquatic organisms on reclaimed mined land. In particular, the Forestry Reclamation Approach (FRA; Zipper *et al.* 2011) encourages reforestation through improving soil conditions and planting native tree species. Through soil reconstruction, the FRA also aids in the restoration of landscape hydrologic patterns and reduction in ion concentrations in water (Sena *et al.* 2014). Techniques advocated through the FRA may restore the microhabitat and microclimatic conditions in both terrestrial and stream environments required by salamanders and other stream biota.

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Data accessibility

Salamander count data and site and sampling covariates available from Dryad Digital Repository doi 10.5061/dryad.5m8f6 (Price *et al.* 2015).

References

- Acton, P.M., Fox, J.F., Campbell, J.E., Jones, A.L., Rowe, H., Martin, D. & Bryson, S. (2011) Role of soil health in maintaining environmental suitability of surface coal mining. *Environmental Science and Technology*, **45**, 10265–10272.
- Allan, J.D. (2004) Landscapes and riverscapes: the influence of land use on stream ecosystems. *Annual Review in Ecology, Evolution and Systematics*, **35**, 257–284.
- Barrett, K. & Price, S.J. (2014) Stream salamanders and urbanization: a review, conservation options and research needs. *Freshwater Science*, **33**, 927–940.
- Barrett, K., Helms, B.S., Samoray, S.T. & Guyer, C. (2010) Growth patterns of a stream vertebrate differ between urban and forested catchments. *Freshwater Biology*, **55**, 1628–1635.
- Bernhardt, E.S. & Palmer, M.A. (2011) The environmental costs of mountaintop mining valley fill operations for aquatic ecosystems of the Central Appalachians. *Annals of New York Academy of Science*, **1223**, 39–57.
- Bridge, G. (2004) Contested terrain: mining and the environment. *Annual Review of Environment and Resources*, **29**, 205–259.
- Chambers, D.L. (2011) Increased conductivity affects corticosterone levels and prey consumption in larval amphibians. *Journal of Herpetology*, **45**, 219–223.
- Chaney, W.R., Pope, P.E. & Byrnes, W.R. (1995) Tree survival and growth on land reclaimed in accord with Public Law 95-87. *Journal of Environmental Quality*, **24**, 630–634.
- Connette, G.M. & Semlitsch, R.D. (2013) Life history as a predictor of salamander recovery rate from timber harvest in southern Appalachian Forests, USA. *Conservation Biology*, **27**, 1399–1409.
- Crawford, J.A. & Semlitsch, R.D. (2007) Estimation of core terrestrial habitat for stream-breeding salamanders and delineation of riparian buffers for protection of biodiversity. *Conservation Biology*, **21**, 152–158.
- Davic, R.B. & Welsh, H.H. Jr (2004) On the ecological roles of salamanders. *Annual Review of Ecology, Evolution and Systematics*, **35**, 405–434.
- Dorazio, R.M., Martin, J. & Edwards, H.H. (2013) Estimating abundance while accounting for rarity, correlated behavior, and other sources of variation in counts. *Ecology*, **94**, 1472–1478.
- Ferrari, J.R., Lookingbill, T.R., McCormick, B., Townsend, P.A. & Eshleman, K.N. (2009) Surface mining and reclamation effects on flood response of watersheds in the central Appalachian Plateau region. *Water Resources Research*, **45**, W04407.
- Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R. *et al.* (2005) Global consequences of land use. *Science*, **309**, 570–574.
- Fox, J.F. (2009) Identification of sediment sources in forested watersheds with surface coal mining disturbance using carbon and nitrogen isotopes. *American Water Resources Association*, **45**, 1273–1289.
- Fritz, K.M., Fulton, S., Johnson, B.R., Barton, C.D., Jack, J.D., Word, D.A. & Burke, R.A. (2010) Structural and functional characteristics of natural and constructed channels draining a reclaimed mountaintop removal and valley fill coal mine. *Journal of the North American Benthological Society*, **29**, 673–689.
- Gelman, A.E., Meng, X. & Stern, H.S. (1996) Posterior predictive assessment of model fitness via realized discrepancies. *Statistica Sinica*, **6**, 733–807.
- Gelman, A.E. & Rubin, D.B. (1992) Inference from iterative simulation using multiple sequences. *Statistical Science*, **7**, 457–511.
- Grant, E.H.C., Nichols, J.D., Lowe, W.H. & Fagan, W.F. (2010) Use of multiple dispersal pathways facilitates amphibian persistence in stream networks. *Proceedings of the National Academy of Sciences, USA*, **107**, 6936–6940.
- Greenberg, A.E., Clesceri, L.S. & Eaton, A.D. (1992) *Standard Methods for the Examination of Water and Wastewater*, 18th edn. American Public Health Association, Washington, DC.
- Greene, B.T., Lowe, W.H. & Likens, G.E. (2008) Forest succession and prey availability influence the strength and scale of terrestrial-aquatic linkages in a headwater salamander system. *Freshwater Biology*, **53**, 2234–2243.
- Griffith, M.B., Norton, S.B., Alexander, L.C., Pollard, A.I. & LeDuc, S.D. (2012) The effects of mountaintop mines and valley fills on the physicochemical quality of stream ecosystems in the central Appalachians: a review. *Science of the Total Environment*, **417**, 1–12.
- Harper, K.A., MacDonald, S.E., Burton, P.J., Chen, J., Brosfoske, K.D., Saunders, S.C. *et al.* (2005) Edge influence of forest structure and composition in fragmented landscapes. *Conservation Biology*, **19**, 768–782.
- Hitt, N.P. & Chambers, D.B. (2014) Temporal changes in taxonomic and functional diversity of fish assemblages downstream from mountaintop mining. *Freshwater Science*, **33**, 915–926.
- Hopkins, R.L. & Roush, J.C. (2013) Effects of mountaintop mining on fish distributions in central Appalachia. *Ecology of Freshwater Fish*, **22**, 578–586.
- Johnson, J.E. & Goldberg, A.S. (1975) Movement of larval two lined salamanders (*Eurycea bislineata*) in the Mill River, Massachusetts. *Copeia*, **1975**, 588–589.
- Karraker, N.E., Gibbs, J.P. & Vonesh, J.R. (2008) Impacts of road deicing salt on the demography of vernal pool-breeding amphibians. *Ecological Applications*, **18**, 724–734.
- Keitzer, S.C. & Goforth, R.R. (2013) Salamander diversity alters stream macroinvertebrate community structure. *Freshwater Biology*, **58**, 2114–2125.
- Kéry, M. & Schaub, M. (2012) *Bayesian Population Analysis Using WinBUGS: A Hierarchical Perspective*. Academic Press, Waltham, Massachusetts.
- Kujala, H., Whitehead, A.L., Morris, W.K. & Wintle, B.A. (2015) Towards a strategic offsetting of biodiversity loss using spatial concepts and tools: a case study on mining impacts in Australia. *Biological Conservation*, **192**, 513–521.
- Lowe, W.H. (2012) Climate change is linked to long-term decline in a stream salamander. *Biological Conservation*, **145**, 48–53.
- Lowe, W.H., Nislow, K.H. & Bolger, D.T. (2004) Stage-specific and interactive effects of sedimentation and trout on a headwater stream salamander. *Ecological Applications*, **14**, 164–172.
- Lunn, D., Spiegelhalter, D., Thomas, A. & Best, N. (2009) The BUGS project: evolution, critique, and future directions. *Statistics in Medicine*, **28**, 3049–3067.

- MacKenzie, D.A., Nichols, J.D., Hines, J.E., Knutson, M.G. & Franklin, A.B. (2003) Estimating site occupancy, colonization and local extinction when a species is detected imperfectly. *Ecology*, **84**, 2200–2207.
- Milanovich, J.R., Maerz, J.C. & Rosemond, A.D. (2015) Stoichiometry and estimates of nutrient standing stocks of larval salamanders in Appalachian headwater streams. *Freshwater Biology*, **60**, 1340–1353.
- Muncy, B., Price, S.J., Bonner, S.J. & Barton, C.D. (2014) Mountaintop removal mining reduces stream salamander occupancy and richness in southeastern Kentucky (USA). *Biological Conservation*, **180**, 115–121.
- Negley, T.L. & Eshleman, K.N. (2006) Comparison of storm-flow responses of surface-mined and forested watersheds in the Appalachian Mountains, U.S.A. *Hydrologic Processes*, **20**, 3467–3483.
- Orser, P.N. & Shure, D.J. (1975) Population cycles and activity patterns of the Dusky salamander, *Desmognathus fuscus fuscus*. *American Midland Naturalist*, **93**, 403–410.
- Palmer, M.A., Bernhardt, E.S., Schlesinger, W.H., Eshleman, K.N., Fouloua-Georgiou, E., Hendryx, M.S. *et al.* (2010) Mountaintop mining consequences. *Science*, **327**, 148–149.
- Peterman, W.E., Crawford, J.A. & Semlitsch, R.D. (2008) Productivity and significance of headwater streams: population structure and biomass of the black-bellied salamander (*Desmognathus quadramaculatus*). *Freshwater Biology*, **53**, 347–357.
- Peterman, W.E. & Semlitsch, R.D. (2013) Fine-scale habitat associations of a terrestrial salamander: the role of environmental gradients and implications for population dynamics. *PLoS One*, **8**, e62184.
- Pimm, S.L. & Raven, P. (2000) Biodiversity: extinction by numbers. *Nature*, **403**, 843–845.
- Pond, G.J., Passmore, M.E., Borsuk, F.A., Reynolds, L. & Rose, C.J. (2008) Downstream effects of mountain top coal mining: comparing biological conditions using family- and genus- level macroinvertebrate bioassessment tools. *Journal of American Benthological Society*, **27**, 717–737.
- Price, S.J., Browne, R.A. & Dorcas, M.E. (2012) Evaluating the effects of urbanisation on salamander abundances using a before-after control-impact design. *Freshwater Biology*, **57**, 193–203.
- Price, S.J., Muncy, B.L., Bonner, S.J., Drayer, A.N. & Barton, C.D. (2015) Data from: Effects of mountaintop removal mining and valley filling on the occupancy and abundance of stream salamanders. *Dryad Digital Repository*, <http://dx.doi.org/10.5061/dryad.5m8f6>.
- Redmond, W.H. (1980) Notes on the distribution and ecology of the black mountain dusky salamander *Desmognathus walteri* Barbour (Amphibia: Plethodontidae) in Tennessee. *Brimleyana*, **4**, 123–131.
- Schmidt, H. & Glaesser, C. (1998) Multitemporal analysis of satellite data and their use in the monitoring of the environmental impacts of open cast lignite mining areas in Eastern Germany. *International Journal of Remote Sensing*, **19**, 2245–2260.
- Sena, K., Barton, C., Angel, P., Agouridis, C. & Warner, R. (2014) Influence of spoil type on chemistry and hydrology of interflow on a surface coal mine in the eastern US coalfield. *Water, Air, & Soil Pollution*, **225**, 2171.
- Simmons, J.A., Currie, W.S., Eshleman, K.N., Kuers, K., Monteleone, S., Negley, J.L., Pohlard, B.R. & Thomas, C.L. (2008) Forest to reclaimed mine land use change leads to altered ecosystem structure and function. *Ecological Applications*, **18**, 104–118.
- Surasinghe, T.D. & Baldwin, R.F. (2015) Importance of riparian forest buffers in conservation of stream biodiversity: responses to land uses by stream-associated salamanders across two southeastern temperate ecoregions. *Journal of Herpetology*, **49**, 83–94.
- US Environmental Protection Agency (2011) The Effects of Mountaintop Mines and Valley Fills on Aquatic Ecosystems of the Central Appalachian Coalfields. USEPA. Report no. EPA/600/R-09/138F.
- Wickham, J., Wood, P.B., Nicholson, M.C., Jenkins, W., Druckenbrod, D., Suter, G.W. *et al.* (2013) The overlooked terrestrial impacts of mountaintop mining. *BioScience*, **63**, 335–348.
- Wiken, E., Jiménez Nava, F. & Griffith, G. (2011) *North American Terrestrial Ecoregions – Level III*. Commission for Environmental Cooperation, Montreal, Canada.
- Wood, P.B. & Williams, J.M. (2013a) Impact of valley fills on streamside salamanders in southern West Virginia. *Journal of Herpetology*, **47**, 119–125.
- Wood, P.B. & Williams, J.M. (2013b) Terrestrial salamander abundance on reclaimed mountaintop removal mines. *Wildlife Society Bulletin*, **37**, 815–823.
- Zipper, C.E., Burger, J.A., Skousen, J.G., Angel, P.N., Barton, C.D., Davis, V. & Franklin, J.A. (2011) Restoring forests and associated ecosystem services on Appalachian coal surface mines. *Environmental Management*, **47**, 751–765.

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Supporting Information

Additional Supporting Information may be found in the online version of this article.

Appendix S1. Examples of stream reaches sampled for stream salamanders on (a) a reclaimed surface mine and (b) reference stream in University of Kentucky's Robinson Forest.

Appendix S2. R and OpenBUGS code for the hurdle model of abundance.

Appendix S3. Results of the sensitivity analysis.