Articles

Responses of Terrestrial Herpetofauna to Persistent, Novel Ecosystems Resulting from Mountaintop Removal Mining

Jennifer M. Williams,* Donald J. Brown, Petra B. Wood

J.M. Williams
West Virginia Cooperative Fish and Wildlife Research Unit, Division of Forestry and Natural Resources, West Virginia University, Morgantown, West Virginia 26506

D.J. Brown
West Virginia University, Division of Forestry and Natural Resources, P.O. Box 6125, Morgantown, West Virginia 26506; U.S. Forest Service, Northern Research Station, P.O. Box 404, Parsons, West Virginia 26287

P.B. Wood
U.S. Geological Survey, West Virginia Cooperative Fish and Wildlife Research Unit, West Virginia University, Morgantown, West Virginia 26506

Abstract

Mountaintop removal mining is a large-scale surface mining technique that removes entire floral and faunal communities, along with soil horizons located above coal seams. In West Virginia, the majority of this mining occurs on forested mountaintops. However, after mining ceases the land is typically reclaimed to grasslands and shrublands, resulting in novel ecosystems. In this study, we examined responses of herpetofauna to these novel ecosystems 10–28 y postreclamation. We quantified differences in species-specific habitat associations, (sub)order-level abundances, and habitat characteristics in four habitat types: reclaimed grassland, reclaimed shrubland, forest fragments in mined areas, and nonmined intact forest. Habitat type accounted for 33.2% of the variation in species-specific captures. With few exceptions, forest specialists were associated with intact forest and fragmented forest sites, while habitat generalists were either associated with grassland and shrubland sites or were distributed among all habitat types. At the (sub)order level, salamander (Order Urodela) captures were highest at fragmented and intact forest sites, frog and toad (Order Anura) captures were lowest at intact forest sites, and snake (Suborder Serpentes) captures were highest at shrubland sites. Habitat type was a strong predictor for estimated total abundance of urodeles, but not for anurans or snakes. Tree stem densities in grasslands differed from the other three habitat types, and large trees (>38 cm diameter at breast height) were only present at forest sites. Overstory vegetation cover was greater in forested than in reclaimed habitat types. Ground cover in reclaimed grasslands was distinct from forest treatments with generally less woody debris and litter cover and more vegetative cover. It is important to consider the distributions of habitat specialists of conservation concern when delineating potential mountaintop mine sites, as these sites will likely contain unsuitable habitat for forest specialists for decades or centuries when reclaimed to grassland or shrubland.

Keywords: Amphibian; Appalachia; disturbance; forest; reptile; restoration; soil; West Virginia

Received: October 18, 2016; Accepted: June 30, 2017; Published Online Early: July 2017; Published: December 2017

Citation: Williams JM, Brown DJ, Wood PB. 2017. Responses of terrestrial herpetofauna to persistent, novel ecosystems resulting from mountaintop removal mining. *Journal of Fish and Wildlife Management* 8(2):387–400; e1944-687X. doi:10.3996/102016-JFWM-079

Copyright: All material appearing in the *Journal of Fish and Wildlife Management* is in the public domain and may be reproduced or copied without permission unless specifically noted with the copyright symbol ©. Citation of the source, as given above, is requested.

The findings and conclusions in this article are those of the author(s) and do not necessarily represent the views of the U.S. Fish and Wildlife Service.

Present address: Partners in Amphibian and Reptile Conservation Federal Coordinator, 1201 Oakridge Drive, Fort

Collins, Colorado 80525

* Corresponding author: 1jenmwilliams@gmail.com

Introduction

Mountaintop removal mining, one type of surface mining in the Appalachian region (the Appalachian mountain area, which runs from southern New York to northern Mississippi, Alabama, and Georgia), transforms landscapes from relatively contiguous, native forests to nonnative grasslands and shrub habitats containing forest fragments (Weakland and Wood 2005; Wood et al. 2006; Wickham et al. 2007). Disturbance associated with this mining technique includes removal of entire floral and faunal communities and loss of soil horizons in the process of excavation (Bernhardt and Palmer 2011), reclamation of mined areas with primarily invasive and exotic vegetation, and arrested patterns of succession (Zipper et al. 2011). These changes in soil and vegetative communities affect abiotic conditions and other factors that lead to creation of a novel ecosystem. Even 28 y postreclamation on a mountaintop removal mine site in West Virginia, the area was still dominated by grasses and shrubs with very little tree growth or canopy cover (Wood and Williams 2013). In contrast, typical patterns of succession for forests in the central Appalachians include trees dominating regeneration within 5 y postdisturbance in clear-cuts; within 10 y, development of tree canopies over 6 m in height is common (Smith 1977).

In the Appalachian region, where surface coal mining is the leading cause of deforestation (Brady 2015), more than 600,000 ha have been surface-mined for coal and more than 10,000 additional hectares are being mined each year (Zipper et al. 2011). Surface coal mining is the leading cause of deforestation in the Appalachian region (Brady 2015). Additionally, biologists have long recognized the Appalachians as a hotspot of salamander biodiversity. The potentially significant impacts of surface coal mining on regional or global herpetofaunal biodiversity merit further exploration by researchers.

Despite the large amount of area subjected to surface mining, we found only three published studies that investigated responses of terrestrial amphibian and reptile communities to mining and postmining reclamation in this region. Lacki et al. (2005) sampled snakes at a bottomland forest corridor and riparian area, consisting mainly of midsuccessional hardwood species and located adjacent to a surface mine. Abundance of copperbelly watersnakes (*Nerodia erythrogaster neglecta*) was higher after mining when compared to premining and duringmining levels. Wood and Williams (2013) found that terrestrial salamander abundance was higher in forested habitat than in reclaimed grassland habitat and reclaimed shrubland habitat. Abundance increased in forests with increasing distance from mine edges. Brady (2015) determined that salamander species richness was higher in unmined reference forests than in abandoned surface mines that had reforested naturally, but that relative abundance did not differ between treatments. However, when Brady excluded stream-dependent species from analyses, salamander richness did not differ. We can draw no strong patterns regarding terrestrial herpetofauna response to mining from these limited studies.

Responses of herpetofauna to superficially similar types of landscape change, such as forest clear-cutting, are comparatively well studied (e.g., Petranka et al. 1993; deMaynadier and Hunter 1995; Duguay and Wood 2002; Ford et al. 2002). However, other types of disturbance are not necessarily a good proxy for predicting how individual species will respond to mountaintop mine reclamation. For example, stumps and some understory vegetation are typically retained on the landscape following clear-cutting, assisting with soil stability, whereas mountaintop removal mining removes the soil in addition to both overstory and understory vegetation. Further, while clear-cutting operations may result in moderately compacted soil from machinery, reclaimed mountaintop mines contain unweathered rock material that is highly compacted to facilitate slope stability (Acton et al. 2011). Amphibians and reptiles can use soil for aestivation, hibernation, refuge, and foraging (Taub 1961; Ernst et al. 1994; Petranka 1998; Bailey et al. 2004; Dodd 2013), and soil conditions typically associated with reclaimed mine surfaces are unlikely to provide suitable habitat for herpetofaunal species that use soils for one or more of these purposes. Most land management practices affect smaller areas and cause less disturbance than mountaintop removal mining. Because of the degree of forest floor vegetation and soil that are removed, mountaintop removal mining can be classified as a severe disturbance, similar to rapid glacier advancement, glacier retreat, or landslides (Oliver and Larson 1996). The novel ecosystems created as a result of mountaintop removal mining, just like other novel ecosystems, can inhibit restoration of the previous system and the way it functioned (Hobbs et al. 2005). We should note that reclamation is not synonymous with restoration. Reclamation targets land or a specific site and has geotechnical stability as its legacy. Conversely, restoration targets ecosystems, aiming to return them to their initial land use and functionality (Lima et al. 2016).

The purpose of this study was to improve our understanding of how herpetofauna respond at the species-specific level and (sub)order levels to habitat characteristics associated with mountaintop mine reclamation. We conducted this study at reclaimed mountaintop removal sites and nearby intact forest reference sites in West Virginia. We investigated three habitat types commonly produced from mine reclamation, including reclaimed grasslands, reclaimed shrublands, and forest

fragments, and used sites that had sufficient time for recolonization by herpetofauna (i.e., 10–28 y postreclamation). To investigate herpetofaunal responses, we assessed differences in species-level habitat associations and estimated abundance among habitat types at the (sub)order level. We anticipated that, excluding times of the year when soil is needed for aestivation, most reptiles would respond favorably to reclaimed habitat, because the higher temperature and insolation on these sites could facilitate longer active temperatures for these ectotherms, and conversely, most amphibians would respond negatively due to their requirement of cool, moist habitat to prevent desiccation. The information in this study will be useful for predicting how herpetofaunal species will respond to reclaimed mine habitats, allowing for reclamation strategies that maximize habitat suitability for species of conservation concern.

Study Site

Our study area included three mountaintop removal mines and nearby intact forest in southwestern West Virginia in Boone, Kanawha, Fayette, and Logan counties. The mines were spatially independent, with distance between mines ranging from ca. 25 to 60 km. Sites were located within the Allegheny Plateau physiographic province and the Appalachian coalfield, and were characterized by moderate to strong relief. Habitat types on the three mines included reclaimed grasslands (1,672, 1,819, and 2,003 ha), reclaimed shrublands (106, 428, and 508 ha), and forest fragments (155, 214, and 339 ha).

We considered forests that were bordered on at least three sides by reclaimed habitat to be forest fragments. Our reference habitat type was proximal intact, mature eastern deciduous forest, which was not located on mined land and was not directly impacted by mining activity.

Fragmented and intact forests contained 60–80-y-old, mature second-growth hardwoods. Overstory species included tuliptree (*Liriodendron tulipifera*), red and sugar maples (*Acer rubrum* and *Acer saccharum*, respectively), American sycamore (*Plantanus occidentalis*), northern red, white, and black oaks (*Quercus rubra, Quercus alba*, and *Quercus velutina*, respectively); pignut, bitternut, and shagbark hickories (*Carya glabra*, *Carya cordiformis*, and *Carya ovata*, respectively); American beech (*Fagus grandifolia*), white ash (*Fraxinus americana*), and black birch (*Betula lenta*). Understory species (seedlings, saplings, and poles) included black gum (*Nyssa sylvatica*), flowering dogwood (*Cornus florida*), ironwood (*Carpinus caroliniana*), spicebush (*Lindera benzoin*), and other common hardwood species, including the aforementioned overstory tree species.

Mined lands reclaimed to grasslands ranged in age from 10 to 18 y (mean = 15 y) postreclamation. They were seeded with grass and forb species, primarily nonnatives that can rapidly colonize a site and provide soil stability. Predominant grasses included tall fescue (*Festuca arundinacea*), orchard grass (*Dactylis glomerata*), and broomsedge (*Andropogon virginicus*; Wood and Ammer 2015). Common legumes included birdsfoot trefoil (*Lotus corniculatus*) as well as sericea and bicolor lespedezas (*Lespedeza cuneata* and *Lespedeza bicolor*, respectively). Grasslands occasionally contained a few planted shrubs, primarily nonnative autumn olive (*Elaeagnus umbellata*) and multiflora rose (*Rosa multiflora*).

Reclaimed shrublands contained shrub, sapling, and pole-sized (<8 cm diameter) stems and were 18–28 y (mean = 23 y) postreclamation. We used vegetation structure and composition to define reclaimed treatments because reclamation age of grasslands and shrublands overlapped. As noted by Wood and Williams (2013), even 28 y postreclamation, reclaimed sites did not exhibit normal patterns of succession; those planted as grasslands remained as grasslands, while those planted as shrublands remained as shrublands (i.e., shrublands were not the result of grasslands that grew over time into shrublands). Predominant shrubland species included autumn olive, multiflora rose, European black alder (*Alnus glutinosa*), blackberry and raspberry (*Rubus* spp.), sourwood (*Oxydendrum arboreum*), black locust (*Robinia pseudoacacia*), scotch and white pines (*Pinus sylvestris* and *Pinus strobus*, respectively), and species found in the surrounding native landscape, such as red maple, American sycamore, and tuliptree (Williams 2003). Most of these species were planted during reclamation, although early successional native species self-seeded. Many of the grass and forb species found in reclaimed grasslands also occurred in reclaimed shrublands.

Thomas et al. (2000) sampled native and reclaimed soils on one of the mines that we studied. They found that average solum (i.e., topsoil and subsoil; Plaster 1996) depth of native soils was 97 cm, and all native soils sampled had O horizons (i.e., organic horizons above mineral soil). In contrast, on reclaimed soils, they found that solum depth ranged from 12 cm (2-y-old reclaimed) to 31 cm (23-y-old reclaimed), and that O horizons developed on some mine soils within 2 y of age, whereas other reclaimed soils up to 11 y old did not have any O horizon. Some young mine soils showed little to no profile development, while in others, the A horizons (i.e., topmost mineral horizons; contain some organic matter) increased with time and became deeper than native soils in as few as 7 y, which Thomas et al. (2000) attributed to the seeding of grasses and legumes during reclamation. Thomas et al. (2000) characterized native soils in the same county where they sampled mountaintop removal mine soils as deep or moderately deep, and Williams (2003) found native soils in the region to have high leaf litter cover.

Methods

Herpetofauna sampling

We sampled herpetofauna with drift fence arrays, using three arrays in each habitat type. Array center points were located at randomly selected points that were positioned 35 m from streams or rip-rap channels, and 75–2,500 m from mine/forest edges to ensure we were sampling the community within each habitat type.

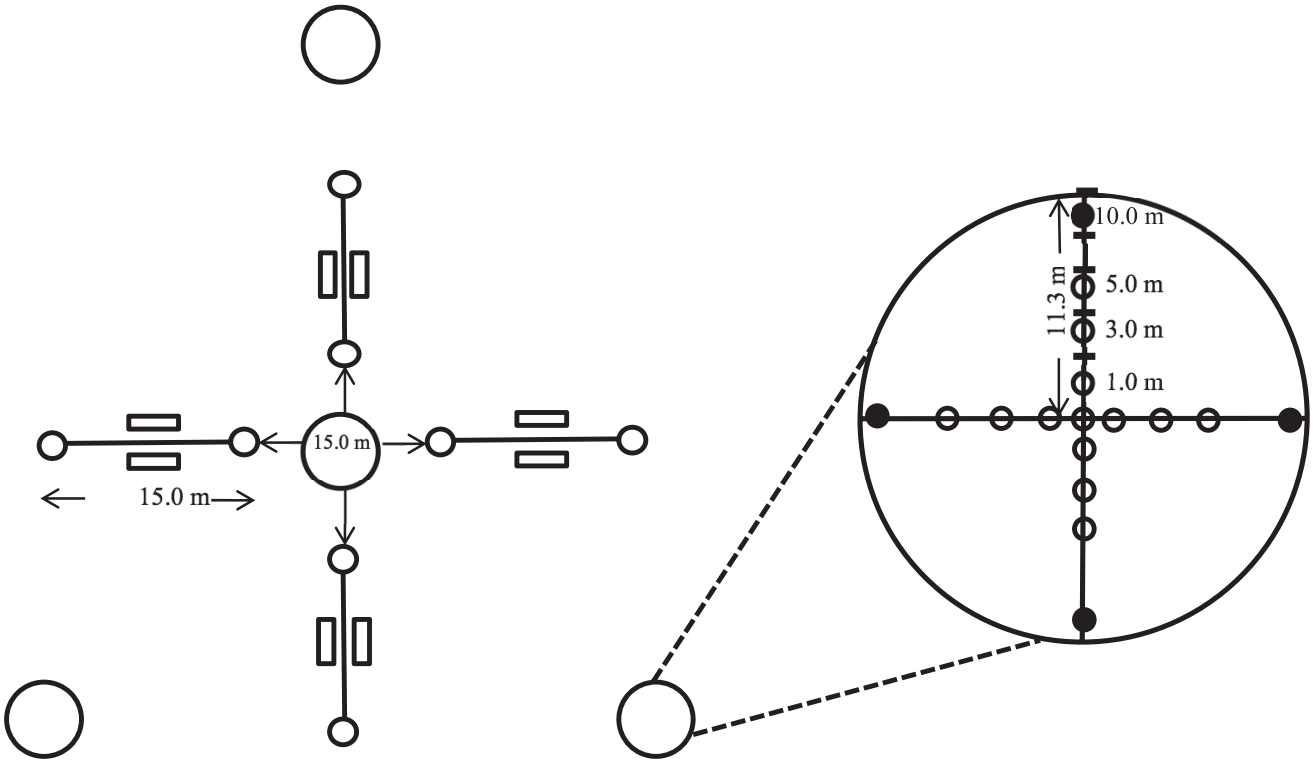

Figure 1. Herpetofaunal drift fence array design. Four vegetation subplots (0.04 ha) were established at each herpetofaunal drift fence array sampling point at the center and 35.0 m away at 0, 120, and 240°. The black solid rectangles represent locations where percentage of ground cover and percentage of canopy cover were measured. The hollow circles represent locations in reclaimed grassland and reclaimed shrubland where vegetative cover (Robel pole index) and vegetation height were measured to the nearest 0.5 dm using a Robel pole, and where organic litter depth measurements were taken to the nearest centimeter using a metric ruler. Herpetofauna were sampled 2000–2002 and habitat was sampled in 2002 in Boone, Kanawha, Fayette, and Logan counties, West Virginia. Figure is not to scale.

Drift fences consisted of 30-cm-tall plastic silt fencing (Enge 1997), with four 15-m arms (Vogt and Hine 1982) supported by wooden stakes (Figure 1). We arranged arms in a plus-shaped design with a central separation of 15 m from the arm directly opposite it (Corn 1994). We positioned pitfalls (18.9-L plastic buckets) at the ends of each arm, totaling eight pitfalls per array, and buried them flush with the soil surface. We placed funnel traps (minnow trap no. 1275; Frabill, Jackson, WI) in the middle on each side of every array arm. We placed soil and litter on edges of funnel trap openings and around rims of buckets. To prevent dehydration of captured animals, we kept wet sponges or paper towels in each pitfall and funnel trap during sampling periods. We provided shade to captures by securing a piece of silt fencing over the funnel traps and by elevating plastic bucket lids approximately 10 cm over the pitfall openings with the use of untreated lumber.

We opened arrays for four consecutive nights in February 2002 and for 8–12 consecutive nights in March and May–October 2000–2002 (with the exception of August 2001 due to logistical constraints), equaling 21 trapping sessions. We checked traps every other day in 2000 and 2001 (Vogt and Hine 1982; Brenner et al. 1992; Corn 1994). In 2002, we checked traps every third day due to logistical constraints. Concurrent sampling of treatments avoided temporal biases. We identified, measured, and marked captured individuals, and then released them at least 3 m from drift fence arrays (Vogt and Hine 1982).

Habitat sampling

We sampled habitat characteristics and vegetation in 2002 using modified methods of James and Shugart (1970) and the Breeding Bird Research Database program (Martin et al. 1997). We established four vegetation subplots (each 0.04 ha) at each herpetofaunal drift fence array sampling point at the center and 35.0 m away at 0, 120, and 240° (Figure 1). Within each 0.04-ha subplot, we identified all tree species, measured diameter at breast height (dbh), and categorized trees into one of two dbh classes: 8.0–38.0 cm and >38.0 cm. We established a 5.0-m radius circle at the center of each vegetation subplot, in which we recorded number of woody stems <8.0 cm dbh.

We measured percentage of ground cover and percentage of canopy cover using an ocular sighting tube on each subplot (James and Shugart 1970) every 2.3 m along each of the four 11.3-m transects that intersected in the subplot center, totaling 20 measure-

ments per subplot. In all treatments, percentage of ground cover included green (grass, shrubs, fern, or herbaceous vegetation <0.5 m in height), vegetative litter, woody debris, moss, and *Lespedeza* spp. We measured *Lespedeza* spp. separately because of the invasiveness of this genus on reclaimed areas. We recorded percentage of canopy cover for four height classes: >0.5–3.0 m, 3.1–6.0 m, 6.1–12.0 m, and >12.0 m. We calculated percentage of cover for each variable by summing the number of points at which each variable was present and dividing by 20.

For reclaimed grassland and shrubland treatments, we measured vegetative cover using a Robel pole at the subplot center (four measurements were taken at each subplot center while facing the center from each transect direction), and at 1.0, 3.0, and 5.0 m along each transect (Robel et al. 1970), for a total of 16 measurements. In these two treatments, we also measured maximum green height (forb and grass height) to the nearest 0.5 dm using the Robel pole at the subplot center and at 1.0, 3.0, 5.0, and 10.0 m along each transect, for a total of 16 measurements. We took maximum height measurements by choosing the tallest forb or grass within a 1.0-dm radius from the specified transect locations. Vegetative cover and vegetation height at a subplot was the average of these 16 measurements. In all treatments, we measured organic litter depth to the nearest centimeter using a metric rule at the subplot center and at 1.0, 3.0, and 5.0 m along each transect, totaling 13 measurements. At each subplot center, we measured percentage of slope using a clinometer and aspect with a compass.

Statistical analyses

We assessed species-level herpetofaunal associations with habitat types using redundancy analysis, which is an extension of principal components analysis (PCA) to include explanatory variables. Specifically, each response variable is regressed on each explanatory variable and then a PCA is performed on the matrix of fitted values (McCune and Grace 2002). We chose redundancy analysis, which assumes response variables are related linearly to predictors, over canonical correspondence analysis, which assumes response variables are related unimodally to predictors, because our predictor was categorical (Lepš and Šmilauer 2003). The response variable was the total number of unique individuals captured for each species at each site during 2000–2002, with the data standardized to create a correlation response matrix (Borcard et al. 2011). To determine if habitat type associations in this study agreed with typical habitat associations for each species, we classified each species as either a forest specialist or a habitat generalist based on whether the species is typically encountered only in forested habitat or is commonly found in both forest and nonforest habitat types, respectively (Table 1; Green and Pauley 1987; Petranka 1998; Lannoo 2005). We tested for global evidence of species–habitat type associations using a permutation test with 1,000 replications ($\alpha = 0.05$). We visually assessed species-level associations using a distance biplot, where the angles between the species data and habitat types reflect their correlations, and the distances among habitat types reflect their Euclidean distance (Borcard et al. 2011). We performed this analysis using the software package vegan (version 2.3-4) in program R.

In addition to assessing species-level–habitat type associations, we estimated abundance of herpetofauna in each habitat type using single-season *N*-mixture models (Kéry and Royle 2016). Due to low captures for most species, we estimated abundance at the (sub)order level for Urodela (salamanders), Anura (frogs and toads), and Serpentes (snakes). We did not include Lacertilia (lizards) or Testudines (turtles) in abundance estimates due to few total captures (Table 1). We restricted these analyses to the 2000–2001 survey period because the frequency of trap-checking differed in 2002, and thus the sampling effort was not equivalent. We did not use multiseason models because our data sets were not robust enough to reliably estimate the additional parameters. Our total sampling effort included 12 sites (three sites per habitat type), each with 54 sampling days.

We analyzed (sub)order abundance data using *N*-mixture models, which use both spatial and temporal replication of count data to jointly estimate abundance and detection probability (P), accounting for observed numbers being a product of both ecological and observational processes (Royle 2004). To determine if habitat type was a strong predictor of (sub)order-level herpetofaunal abundance, we used a model selection approach with Akaike's information criterion corrected for small sample size (AICc; Burnham et al. 2011). We also tested if day of year was a strong observation covariate for P, to account for seasonality differences in activity patterns if necessary. Preliminary analyses indicated that for our data sets, this predictor was best modeled as a linear, rather than quadratic, function. For each (sub)order, we ranked four *N*-mixture model structures that represented our two covariates of interest and corresponding null models (Table 2), and selected the model structure with the highest model weight as the top model. We initially intended to include additional variables in the model selection to assess the importance of individual habitat characteristics, but low site replication ($n = 12$) resulted in lack of convergence when the categorical variable habitat type was replaced with continuous variables.

For all models, we used a binomial distribution for the observation (i.e., detection) process, and a Poisson distribution for the state (i.e., abundance) process after comparing AIC values and residual diagnostic plots for three distributions (i.e., Poisson, zero-inflated Poisson, and negative binomial; Kéry and Royle 2016). We assessed model goodness-of-fit using a parametric bootstrap of the Pearson χ^2 statistic (Mazerolle 2016).

Table 1. Names, species code, typical habitat associations: forest species (F) or habitat generalist (G), and total unique captures of species in this study investigating differences in herpetofaunal abundance and diversity in a central Appalachian mountaintop removal mining landscape representing four habitat types: intact forest (IN), forest fragments (FR), reclaimed shrubland (SH), and reclaimed grassland (GR). Herpetofauna were sampled 2000–2002 in Boone, Kanawha, Fayette, and Logan counties, West Virginia.

Scientific name	Common name	Species code	Habitat	IN	FR	SH	GR
Order Anura (frogs and toads)							
Anaxyrus americanus	American toad	ANAM	G	44	11	45	15
Anaxyrus fowleri	Fowler's toad	ANFO	G	0	0	0	3
Hyla chrysoscelis	Cope's gray treefrog	HYCH	F	0	0	4	0
Lithobates catesbeianus	American bullfrog	LICA	G	1	5	7	3
Lithobates clamitans	Green frog	LICL	G	7	69	55	55
Lithobates palustris	Pickerel frog	LIPA	G	22	76	43	50
Lithobates pipiens	Northern leopard frog	LIPI	G	2	8	9	3
Lithobates sylvaticus	Wood frog	LISY	F	5	2	0	0
Pseudacris crucifer	Spring peeper	PSCR	F	2	1	1	0
Unknown species	—	—	—	15	6	13	9
Total	—	—	—	98	178	177	138
Order Urodela (salamanders)							
Ambystoma maculatum	Spotted salamander	AMMA	F	1	3	2	2
Desmognathus fuscus	Northern dusky salamander	DEFU	F	0	1	0	0
Desmognathus monticola	Seal salamander	DEMO	F	1	2	0	0
Eurycea cirrigera	Southern two-lined salamander	EUCI	F	0	2	0	0
Gyrinophilus porphyriticus	Spring salamander	GYPO	F	0	1	0	0
Hemidactylium scutatum	Four-toed salamander	HESC	F	0	1	0	1
Notophthalmus v. viridescens	Red-spotted newt	NOVI	F	30	42	17	24
Plethodon cinereus	Eastern red-backed salamander	PLCI	F	6	2	0	0
Plethodon glutinosus	Northern slimy salamander	PLGL	F	7	9	2	0
Plethodon kentucki	Cumberland Plateau salamander	PLKE	F	17	2	0	0
Pseudotriton ruber	Northern red salamander	PSRU	F	2	0	0	0
Total	—	—	—	64	65	21	27
Order Squamata, Suborder Lacertilia (lizards)							
Plestiodon fasciatus	Common five-lined skink	PLFA	G	4	4	2	0
Scincella lateralis	Little brown skink	SCLA	F	1	0	0	1
Sceloporus undulatus	Eastern fence lizard	SCUN	G	0	0	2	4
Total	—	—	—	5	4	4	5
Order Squamata, Suborder Serpentes (snakes)							
Agkistrodon contortrix	Northern copperhead	AGCO	G	8	5	14	2
Carphophis amoenus	Eastern wormsnake	CAAM	F	4	0	0	0
Coluber constrictor	North American racer	COCO	G	0	0	37	19
Diadophis punctatus	Ring-necked snake	DIPU	F	1	0	0	0
Heterodon platirhinos	Eastern hog-nosed snake	HEPL	G	0	0	4	1
Lampropeltis triangulum	Eastern milksnake	LATR	G	2	7	5	6
Nerodia sipedon	Northern watersnake	NESI	G	0	0	1	1
Pantherophis spiloides	Gray ratsnake	PASP	G	1	2	6	5
Storeria occipitomaculata	Northern red-bellied snake	STOC	F	1	1	0	0
Thamnophis sirtalis	Common gartersnake	THSI	G	10	14	7	7
Unknown species	—	—	—	0	0	0	1
Total	—	—	—	27	29	74	42
Testudines (turtles)							
Terrapene carolina	Eastern box turtle	TECA	G	1	2	0	0
Total	—	—	—	1	2	0	0

The goodness-of-fit test indicated the Anura and Urodele data sets were overdispersed (i.e., variance $>$ mean), and we accounted for this by inflating the estimated standard errors and 95% confidence intervals (CI) based on the c-hat values (Kéry and Royle 2016). These analyses were conducted using the software packages unmarked (version 0.11-0; Fiske and Chandler 2011), AHM (version 0.01), and AICcmodavg (version 2.0-4) in program R (version 3.2.4).

To quantify differences in vegetative characteristics among treatments, we used analysis of variance (ANOVA). For each habitat variable tested, we specified habitat type as the independent variable and blocked for site. We transformed percentage variables using the arcsine–square root transformation, and stem count variables using the log transformation (Zar 1999). We converted aspect measured in degrees (A) to an aspect code (A′) using the formula $A' = (\mathrm{COS}\ [45 - A] + 1)$, where northeastern facing slopes receive a value of $A' = 2$, reflecting mesic conditions, while southwestern exposures receive a value of $A' = 0$ and reflect xeric conditions (Beers et al. 1966). We used Sheffe's test for

Table 2. Model selection using Akaike's information criterion corrected for small sample size (AICc) to determine the importance of habitat type (Habitat) as a predictor of abundance (λ) and day of year (DOY) as a predictor of detection probability (P) for Anura (frogs and toads), Urodela (salamanders), and Serpentes (snakes) sampled in a central Appalachian mountaintop removal mining landscape representing four habitat types: intact forest (IN), fragmented forest (FR), reclaimed shrubland (SH), and reclaimed grassland (GR). Null model structures for λ and P are shown as (.). Herpetofauna were sampled 2000–2002 in Boone, Kanawha, Fayette, and Logan counties, West Virginia.

Order	Model	Δ AICc	Weight
Anura	λ (.), P (DOY)	0.00	0.91
	λ (Habitat), P (DOY)	4.68	0.09
	λ (Habitat), P (.)	51.22	0.00
	λ (.), P (.)	51.97	0.00
Urodela	λ (Habitat), P (.)	0.00	0.70
	λ (.), P (DOY)	3.40	0.13
	λ (.), P (.)	3.80	0.11
	λ (Habitat), P (DOY)	4.81	0.06
Squamata, Suborder Serpentes	λ (.), P (.)	0.00	0.85
	λ (.), P (DOY)	3.67	0.14
	λ (Habitat), P (.)	7.75	0.02
	λ (Habitat), P (DOY)	16.55	0.00

vegetation variables to compare means among treatments when *F*-tests from the ANOVAs indicated significant differences among treatments. We analyzed data using SAS 8.1 (SAS Institute 2000), with $\alpha = 0.10$. We chose to use $\alpha = 0.10$ rather than the typical $\alpha = 0.05$ to reduce the probability of a type II error due to small sample size (i.e., low statistical power; Caughley and Gunn 1995).

Results

The redundancy analysis indicated significant species–habitat associations ($P = 0.024$), with the habitat type predictor accounting for 33.2% of the variation in standardized species data. The biplot showed that, with few exceptions, forest species were associated with intact forest sites and fragmented forest sites, and generalist species were either associated with grassland and shrubland sites or were distributed among all habitat types (Figure 2). Salamander species were generally detected only in forested habitat, with the exceptions of red-spotted newts (*Notophthalmus v. viridescens*) and spotted salamanders (*Ambystoma maculatum*), which were captured in all habitat types. Anuran species were generally detected in all habitat types. However, wood frogs (*Lithobates sylvaticus*) were only captured in forested habitat and Cope's gray treefrogs (*Hyla chrysoscelis*) were found only in reclaimed shrubland treatments; all individuals of the latter species were captured at the same sampling point. Snakes were generally detected in all habitat types, but captures were typically higher in nonforested habitat. However, common gartersnakes (*Thamnophis sirtalis*) appeared to be more abundant in forested habitat. Species-specific counts by habitat type are provided in Table 1.

For Urodela, we captured 11 species and 36, 37, 11, and 14 individuals in intact forest, fragmented forest, grassland, and shrubland, respectively, during 2000–2001 (Data S1; *Supplemental Material*). The AICc model selection indicated habitat type was a strong predictor of order-level abundance, and day of year did not have a strong influence on P (Table 2). Mean estimated abundance of Urodeles was over twice as high in forested than non-forested habitat types (Table 3). For Anura, we captured 9 species and 52, 101, 112, and 119 individuals in intact forest, fragmented forest, grassland, and shrubland, respectively, during 2000–2001 (Data S1; *Supplemental Material*, http://dx.doi.org/10.3996/102016-JFWM-079.S1). The AICc model selection indicated habitat type was not a strong predictor of order-level abundance, and that day of year influenced P (Figure 3; Table 3). For Serpentes, we captured nine species and 19, 19, 28, and 55 individuals in intact forest, fragmented forest, grassland, and shrubland sites, respectively, during 2000–2001 (Data S1; *Supplemental Material*, http://dx.doi.org/10.3996/102016-JFWM-079.S1). The AICc model selection indicated that habitat type was not a strong predictor of suborder-level abundance, and day of year did not have a strong influence on P (Table 2).

All categories of stem densities differed among habitat types (Table 4). Fewer small (<8.0 cm dbh) trees were found in reclaimed grasslands than in intact forests and shrublands. Reclaimed grasslands also contained fewer midsized (8.0–38.0 cm dbh) trees than the other three habitat types. No reclaimed sites had large trees (>38.0 cm dbh). Litter cover was lower in grasslands than forested treatments, but litter depth was similar among treatments. Woody debris cover was higher in intact forests than in reclaimed treatments. Mean percentage of cover by *Lespedeza* spp. was 38.1% and 35.5% at grassland and shrubland sites, respectively, and 0% at forested sites. Grasslands had lower canopy cover by shrubs (>0.5–3.0 m) and saplings (>3.0–6.0 m) than the other three treatments. Cover from understory (>6.0–12.0 m) and overstory (>12.0 m) vegetation in forested treatments exceeded that in grasslands and shrublands. Other habitat variables measured did not differ among treatments.

Discussion

In our quantification of differences in herpetofaunal communities and habitat characteristics in this mountaintop mining landscape, most notable among the results is the dissimilarity in herpetofaunal species composition between forested and reclaimed treatments. We found habitat specialists in forested treatments, whereas reclaimed treatments were dominated by habitat generalists. Our captures included species categorized as Priority 1 (West Virginia Division of Natural Resources 2015), which are those that are the primary focus for conservation activities in the state of West Virginia. They included northern leopard frogs (*Litho-*

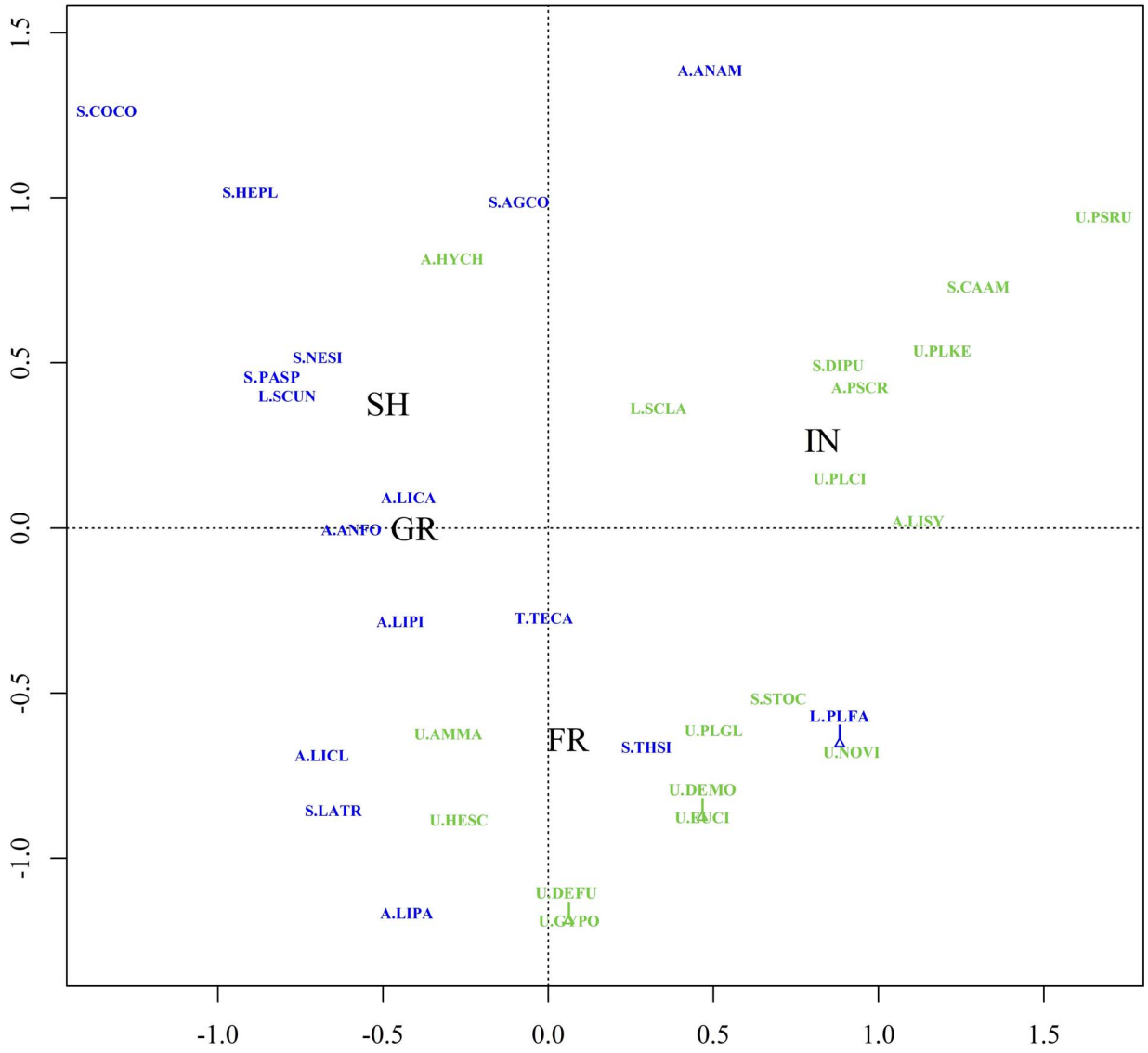

Figure 2. Distance biplot showing species-habitat associations in four central Appalachian mountaintop removal mining landscape habitat types: intact forest (IN), forest fragment (FR), reclaimed shrubland (SH), and reclaimed grassland (GR). Herpetofauna were sampled 2000–2002 and habitat was sampled in 2002 in Boone, Kanawha, Fayette, and Logan counties, West Virginia. The biplot was derived from a redundancy analysis using standardized species data. Species that are forest specialists are shown in green, and species that are habitat generalists are shown in blue. Species acronyms are prefaced with their (sub)order (i.e., Anura [A], Urodela [U], Lacertilia [L], Serpentes [S], and Testudines [T]). Anurans include American toad (*Anaxyrus americanus*; ANAM), Fowler's toad (*Anaxyrus fowleri*; ANFO), Cope's gray treefrog (*Hyla chrysoscelis*; HYCH), American bullfrog (*Lithobates catesbeianus*; LICA), green frog (*Lithobates clamitans*; LICL), pickerel frog (*Lithobates palustris*; LIPA), northern leopard frog (*Lithobates pipiens*; LIPI), wood frog (*Lithobates sylvaticus*, LISY), and spring peeper (*Pseudacris crucifer*; PSCR). Urodeles include spotted salamander (*Ambystoma maculatum*; AMMA), northern dusky salamander (*Desmognathus fuscus*; DEFU), seal salamander (*Desmognathus monticola*; DEMO), southern two-lined salamander (*Eurycea cirrigera*, EUCI), spring salamander (*Gyrinophilus prphyriticus*; GYPO), four-toed salamander (*Hemidactylium scutatum*; HESC), red-spotted newt (*Notophthalmus v. viridescens*; NOVI), eastern red-backed salamander (*Plethodon cinereus*; PLCI), northern slimy salamander (*Plethodon glutinosus*; PLGL), Cumberland Plateau salamander (*Plethodon kentucki*; PLKE), and northern red salamander (*Pseudotriton ruber*; PSRU). Members of Lacertilia included common five-lined skink (*Plestiodon fasciatus*; PLFA), little brown skink (*Scincella lateralis*; SCLA), and eastern fence lizard (*Sceloperus undulatus*; SCUN). Members of the suborder Serpentes include northern copperhead (*Agkistrodon contortrix*; AGCO), eastern wormsnake (*Carphophis amoenus*; CAAM), North American racer (*Coluber constrictor*; COCO), ring-necked snake (*Diadophis punctatus*; DIPU), eastern hog-nosed snake (*Heterodon platirhinos*; HEPL), eastern milksnake (*Lampropeltis triangulum*; LATR), northern watersnake (*Nerodia sipedon*; NESI), eastern ratsnake (*Pantherophis alleghaniensis*; PAAL), northern red-bellied snake (*Storeria occipitomaculata*; STOC), and common gartersnake (*Thamnophis sirtalis*; THSI). Testudines species included eastern box turtle (*Terrapene carolina*; TECA).

Table 3. Mean estimated abundance based on the top model in Table 2 of Anura (frogs and toads), Urodela (salamanders), and Serpentes (snakes) in a central Appalachian mountaintop removal mining landscape representing four habitat types: intact forest (IN), fragmented forest (FR), reclaimed shrubland (SH), and reclaimed grassland (GR). Standard errors (SE) and 95% confidence intervals (CI) represent inflated estimates to account for model overdispersion (i.e., variance > mean). Herpetofauna were sampled 2000–2002 in Boone, Kanawha, Fayette, and Logan counties, West Virginia.

Order	Habitat	Mean abundance	SE	95% CI
Anura	IN	15.8	8.7	5.4–46.6
	FR	30.7	15.5	11.5–82.4
	SH	35.0	17.4	13.2–92.6
	GR	35.0	16.9	13.6–90.0
Urodela	IN	77.2	27.5	38.4–155.1
	FR	78.4	26.2	40.7–150.9
	SH	30.5	14.1	12.3–75.4
	GR	24.2	11.8	9.3–63.1
Squamata, Suborder Serpentes	IN	2.1	1.3	0.6–7.1
	FR	2.3	1.5	0.7–8.1
	SH	6.8	3.5	2.5–18.4
	GR	3.3	1.8	1.1–9.7

bates pipiens) and eastern hog-nosed snakes (*Heterodon platirhinos*), which were associated with reclaimed habitats, and eastern box turtles (*Terrapene carolina*), which were associated with fragmented forests. A fourth Priority 1 species incidentally sighted was the timber rattlesnake (*Crotalus horridus*; Williams 2003). We saw three individuals, one each in fragmented forest, shrubland, and intact forest. Our captures also included three S3 species, a classification that means "vulnerable in the nation or state/province due to a restricted range, relatively few populations (often 80 or fewer), recent and widespread declines, or other factors making it vulnerable to extirpation" (West Virginia Division of Natural Resources 2015; NatureServe 2015). All S3 species were forest specialists that associated with intact forests (Table 1): Cumberland Plateau salamander (*Plethodon kentucki*), northern red salamander (*Pseudotriton ruber*), and eastern wormsnake (*Carphophis amoenus*). These Priority 1 and S3 species showed a range of potential responses, lending further credence to other studies that concluded that responses to mining can be species-specific.

High site fidelity, small home ranges, physiological limitations, low fecundity, and inability to quickly traverse large distances make urodeles especially susceptible to effects of forest alterations (Pough et al. 1987; Petranka et al. 1993; Blaustein et al. 1994). Urodele detection in forested habitats in our study is consistent with observations in studies of impacts to salamanders in clear-cuts. Several habitat features in the forested treatments favored habitation by salamanders, including more trees in larger dbh classes (>8.0 cm) and greater canopy cover from saplings, understory, and overstory when compared to one or both reclaimed treatments. Pough et al. (1987) reported that aboveground foraging activity is reduced for salamanders when understory

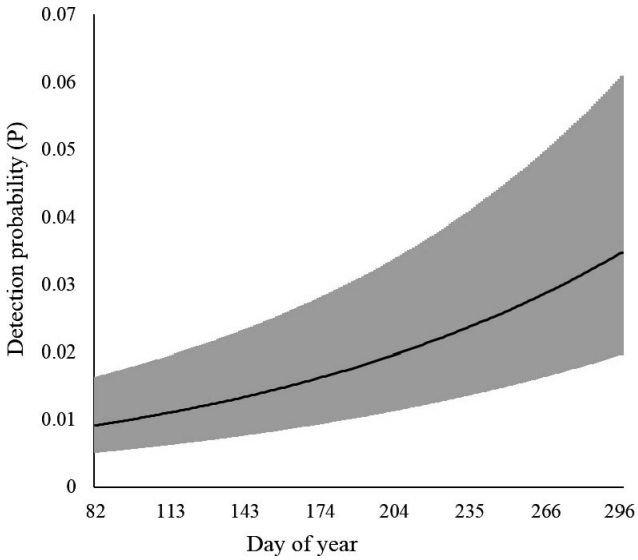

Figure 3. Relationship between day of year and detection probability (P) for anurans (frogs and toads) in a central Appalachian mountaintop removal mining landscape. The black line shows predicted P between days 82 and 297, and the gray shading represents the 95% confidence interval. Day of year was not a strong predictor of P for urodeles (salamanders) or Serpentes species (snakes). Herpetofauna were sampled 2000–2002 in Boone, Kanawha, Fayette, and Logan counties, West Virginia.

vegetation is reduced, which could result in a reduction in biomass production of salamanders and their availability as prey. Microclimate features important to urodeles may be less suitable in reclaimed treatments. Tree removal typically corresponds to a decrease in humidity and soil moisture and an increase in soil temperature (Waldick 1997), factors that can cause desiccation in urodeles. Waldick (1997) reported that quality of leaf litter decreased with conversion of forested habitat to early successional habitats because it often becomes drier in early successional habitats. In our study, grasslands had lower leaf litter cover than all other treatments, and although total cover by leaf litter did not differ between shrublands and forested treatments, the quality of leaf litter may have differed. Leaf litter is important for salamanders because it harbors prey items and provides refuge from hot and dry conditions (Bury 1983; deMaynadier and Hunter 1995). Woody cover was higher in intact forests than in reclaimed treatments. Both woody cover and leaf litter provide sites for salamander species to deposit eggs and are habitat features with which salamanders frequently associate (Green and Pauley 1987). Finally, compaction of mine soils could have rendered them unsuitable for burrowing.

Grant et al. (2016) concluded that there are multiple drivers affecting continental-scale amphibian declines and that more emphasis on local solutions must be implemented to reverse these trends. Because surface coal mining is the primary source of land use change in central Appalachia (Townsend et al. 2009), it may be significantly contributing to regional declines of amphibians, particu-

Table 4. Mean and standard error (SE) for habitat variables used to investigate differences in habitat in a central Appalachian mountaintop removal mining landscape representing four habitat types: intact forest (IN), fragmented forest (FR), reclaimed shrubland (SH) and reclaimed grassland (GR). Dashes indicate treatments where variables were not measured. Within a row, means with the same letters were not different at $\alpha = 0.10$. Data in Table 4 were originally published in Wildlife Society Bulletin 37(4):815–823; 2013; DOI: 10.1002/wsb.319 (Wood and Williams 2013). Habitat was sampled in 2002 in Boone, Kanawha, Fayette, and Logan counties, West Virginia.

Variables	IN (mean ± SE)	FR (mean ± SE)	SH (mean ± SE)	GR (mean ± SE)	*F*	*P*
Slope (%)	34.4 ± 5.3	40.1 ± 5.0	11.4 ± 4.4	22.4 ± 5.2	3.15	0.11
Aspect code	1.0 ± 0.2	1.2 ± 0.3	1.3 ± 0.2	1.1 ± 0.2	0.05	0.98
Litter depth (cm)	3.6 ± 0.7	3.4 ± 0.5	3.8 ± 0.6	2.5 ± 0.4	0.21	0.89
Robel pole index	—	—	4.3 ± 0.7	2.2 ± 0.7	0.27	0.66
Green height (dm; forb and grass)	—	—	8.7 ± 0.9	6.7 ± 0.6	1.34	0.37
Stem counts (no./plot)						
<8.0 cm	2694 ± 519 A	814 ± 213 AB	2732 ± 861 A	228 ± 128 B	5.90	0.03
8.0–38.0 cm	375 ± 43 A	327 ± 44 A	363 ± 102 A	0 ± 0 B	69.80	<0.0001
>38.0 cm	23 ± 8 A	45 ± 13 A	0 ± 0 B	0 ± 0 B	149.46	<0.001
Ground cover (%)						
Litter	69.6 ± 2.6 A	67.7 ± 6.6 A	48.5 ± 7.4 AB	20.4 ± 4.6 B	9.39	0.01
Woody debris	2.9 ± 1.0 A	2.3 ± 1.8 AB	0.0 ± 0.0 B	0.0 ± 0.0 B	3.58	0.09
Moss	1.7 ± 0.9	3.2 ± 1.4	3.5 ± 1.5	0.0 ± 0.0	0.98	0.46
Green	14.2 ± 1.7 B	21.8 ± 6.6 B	47.5 ± 7.8 AB	72.9 ± 6.6 A	10.01	0.009
Lespedeza	0.0 ± 0.0 B	0.0 ± 0.0 B	35.5 ± 9.5 A	38.1 ± 10.8 A	4.68	0.05
Canopy cover (%)						
0.5–3.0 m	35.0 ± 4.5 A	36.4 ± 5.3 A	39.0 ± 6.0 A	0.8 ± 0.6 B	24.48	0.0009
>3.0–6.0 m	52.1 ± 6.8 A	64.6 ± 5.2 A	42.0 ± 7.7 A	0.0 ± 0.0 B	16.20	0.003
6.0–12.0 m	68.3 ± 4.1 A	70.5 ± 6.0 A	12.0 ± 6.6 B	0.0 ± 0.0 B	23.05	0.001
>12.0 m	47.4 ± 4.3 A	48.8 ± 8.6 A	0.0 ± 0.2 B	0.0 ± 0.0 B	69.81	<0.0001

larly salamanders (Wood and Williams 2013; Brady 2015). Previous studies have found it takes decades for salamander populations to recover from clear-cutting (e.g., 15–24 y [Ash 1997; Duguay and Wood 2002], 40–80 y [Crawford and Semlitsch 2008], 50–70 y [Petranka et al. 1993]). In another study in which harvest methods were not specified, it was estimated that ≥100 y were needed for salamander abundance to reach preharvest levels (Connette and Semlitsch 2013). Because mountaintop removal mining is a greater magnitude of disturbance than clear-cutting, recovery times in reclaimed habitats are likely considerably longer (Wood and Williams 2013).

We did not find that habitat type was a strong predictor of total anuran abundance. Anurans possess higher operating and tolerance temperatures than urodeles (Stebbins and Cohen 1995), potentially explaining why many (i.e., 5 of 9) anuran species were detected in all habitat types. Pais et al. (1988) found American toads (*Anaxyrus americanus*) to be associated with dense herbaceous cover in wildlife clearings. Ross et al. (2000) found anuran richness to have a positive relationship with increases in tree basal area. They also observed positive associations between *Lithobates* spp. and woody debris within stands. In our study, total anuran captures were lowest in intact forests, which is where woody debris was highest. Shrubland was the only treatment in which Copes's gray treefrog was found, which is not unusual because this species can be associated with dry and dry-to-mesic northern hardwoods (Dodd 2013), and there was a wetland within close proximity to the sampling point where this species was found. Alternately, wood frogs are associated with moist, deciduous forests (Green and Pauley 1987), so their restriction to forested habitats was expected.

Snakes were generally detected in all habitat types, but most species had higher captures in reclaimed treatments, most notably shrublands. Another study reviewed reptile recolonization of 3–20-y-old postmining restoration sites (Triska et al. 2016). Two Serpentes species were captured in the study, and their detection did not differ between intact reference sites and postmining restoration sites, despite the extra efforts taken to reclaim the mined site (e.g., replacement of topsoil, reestablishment of vegetation from the topsoil seed bank and other sources). Enge and Marion (1986) found more snake captures and twice the snake biomass in a minimum-treatment clear-cut when compared to maximum-treatment clear-cut and a reference treatment (i.e., naturally regenerated 40-y-old slash pine forest; minimum and maximum refer to intensity of harvest and degree of site preparation). Pike et al. (2011) found that in rock outcrops where overstory trees were removed, abundance of two open-habitat specialist Serpentes species increased.

A ubiquitous snake species, North American racer (*Coluber constrictor*), dominated captures in our study, accounting for 33% of total snake captures, and most were in shrublands. Florida kingsnakes (*Lampropeltis getula floridana*) benefited from conversion of native habitat to sugarcane fields due to increased prey density in the sugarcane fields and additional shelter provided by limestone dredge material along banks of irrigation canals (Pough et al. 2001). Riprap channels and rock chimneys in reclaimed mountaintop removal mine habitat may similarly be providing shelter to snakes. Further, Chamblin (2002) found higher small mammal abundance in reclaimed treatments at the same array locations used for this study, which may be attracting a species like the North American racer, an opportunistic feeder (Green and

Pauley 1987). Ross et al. (2000) found snake abundance to be inversely related to tree basal area; similarly, our reclaimed shrubland treatment had low numbers of large tree stems and high snake abundance.

Management implications

Reclamation approaches to speed landscape recovery could benefit salamander conservation. Reclamation scientists recently developed a Forest Reclamation Approach (FRA) that improves on traditional reclamation practices (Zipper et al. 2011). The FRA provides best management practices to facilitate reestablishment of native forests and their accompanying ecosystem services (Zipper et al. 2011). Recommendations include using $\geq$1.2 m of topsoil, weathered sandstone, or similar substrate that is uncompacted and loosely graded to provide a suitable rooting medium, and creating holes deep enough to accommodate root systems of seedlings. The FRA advocates planting herbaceous vegetation with reduced water and nutrient demands and that is short in stature (e.g., bunch-forming grasses, native warm-season grasses) to reduce competition with planted trees and plant colonizers. Further, it is beneficial to plant early successional tree species for wildlife and soil stability, and long-lived tree species characteristic of native deciduous forests in the region. Finally, the FRA also recommends application of fertilizers low in nitrogen and with sufficient phosphorous to support tree growth, (Zipper et al. 2011). The $>$600,000 ha of surface-mined land in the Appalachian region (Zipper et al. 2011) could be targeted for forest reclamation with the FRA as a potential mitigation strategy.

Supplemental Material

Please note: The Journal of Fish and Wildlife Management is not responsible for the content or functionality of any supplemental material. Queries should be directed to the corresponding author.

Data S1. Data used for *N*-mixture model analyses to estimate differences in abundance of herpetofauna. Included in the file are capture data taken in 2000-2001 for Anura, Serpentes, and Urodela (each in a separate worksheet) from drift fence sampling in a central Appalachian mountaintop removal mining landscape in four habitat types: intact forest (IN), fragmented forest (FR), reclaimed shrubland (SH), and reclaimed grassland (GR). The column headings C1 to C54 indicate the number of captures for each of 12 sites (one per row) on each of the 54 sampling days. Data used in the day of year detection probability covariate is in the worksheet labeled Day of Year.

Found at DOI: http://dx.doi.org/10.3996/102016-JFWM-079.S1 (21.2 KB XLSX).

Reference S1. Chamblin HD. 2002. Small mammal communities on a reclaimed mountaintop mine valley fill landscape in southern West Virginia. Master's thesis. Morgantown, West Virginia: West Virginia University.

Found at DOI: http://dx.doi.org/10.3996/102016-JFWM-079.S2 (525 KB PDF).

Reference S2. Grant EHC et al. 2016. Quantitative evidence for the effects of multiple drivers on continental-scale amphibian declines. Scientific Reports 6, Article Number 25625.

Found at DOI: http://dx.doi.org/10.3996/102016-JFWM-079.S3; also available at http://www.nature.com/articles/srep25625 (1283 KB PDF).

Reference S3. Lepš J, Šmilauer P. 2003. Multivariate analysis of ecological data using CANOCO. New York: Cambridge University Press.

Found at DOI: http://dx.doi.org/10.3996/102016-JFWM-079.S4 (5900 KB PDF).

Reference S4. Martin TE, Paine C, Conway CJ, Hockachka WM, Allen P, Jenkins W. 1997. BBIRD Field Protocol. Missoula, Montana: University of Montana.

Found at DOI: http://dx.doi.org/10.3996/102016-JFWM-079.S5; also available at http://www.umt.edu/bbird/docs/BBIRDPROT.pdf (895 KB PDF).

Reference S5. Mazerolle MJ. 2016. Package 'AICcmodavg': model selection and multimodel inference based on (Q)AIC(c). R package version 2.0-4.

Found at DOI: http://dx.doi.org/10.3996/102016-JFWM-079.S6; also available at https://cran.r-project.org/web/packages/AICcmodavg/ (642 KB PDF).

Reference S6. NatureServe. 2015. National and subnational conservation status definitions.

Found at DOI: http://dx.doi.org/10.3996/102016-JFWM-079.S7; also available at http://explorer.natureserve.org/nsranks.htm (109 KB PDF).

Reference S7. Smith HC. 1977. Height of the tallest saplings in 10-year-old Appalachian hardwood clear-cuts. Upper Darby, Pennsylvania: U.S. Forest Service. Research Paper N3-381.

Found at DOI: http://dx.doi.org/10.3996/102016-JFWM-079.S8; also available at http://www.fs.fed.us/ne/newtown_square/publications/research_papers/pdfs/scanned/ne_rp381p.pdf (1428 KB PDF).

Reference S8. Vogt RC, Hine RL. 1982. Evaluation of techniques for assessment of amphibian and reptile populations in Wisconsin. Pages 201–217 in Scott NJ, editor. Herpetological communities: a symposium of the Society for the Study of Amphibians and Reptiles and the Herpetologists' League, August 1977. U.S. Fish and Wildlife Service Resource Report No. 13.

Found at DOI: http://dx.doi.org/10.3996/102016-JFWM-079.S9; also available at https://babel.hathitrust.org/cgi/pt?id=uc1.31822010336709;view=1up;seq=21 (3070 KB PDF).

Reference S9. West Virginia Division of Natural Resources. 2015. 2015 West Virginia State Wildlife Action

Plan. South Charleston, West Virginia: West Virginia Division of Natural Resources.

Found at DOI: http://dx.doi.org/10.3996/102016-JFWM-079.S10; also available at http://www.wvdnr.gov/DRAFT%202015%20WV%20State%20Wildlife%20Action%20Plan%20R1.pdf (19,217 KB PDF).

Reference S10. Williams JM. 2003. Impacts on terrestrial and streamside herpetofauna by mountaintop removal mining in southern West Virginia. Doctoral dissertation. Morgantown, West Virginia: West Virginia University.

Found at DOI: http://dx.doi.org/10.3996/102016-JFWM-079.S11 (3147 KB PDF).

Acknowledgments

We would like to thank Jim Haas (Associate Editor at the Journal of Fish and Wildlife Management), Dr. Kyle Barrett, and two anonymous reviewers for their thoughtful input and careful editing of this manuscript. Research funding was provided by the U.S. Environmental Protection Agency, West Virginia State Legislature, West Virginia Division of Natural Resources, U.S. Geological Survey Cooperative Research Units, and West Virginia University Research Corporation. For field assistance, we thank A. Carroll, S. Bacharach, M. Balcerzak, H.D. Chamblin, T. Dellinger, J. Homyack, J. Martin, J. Simmons, B. Sparks, G. Williams, and J. Woolard. The late Dr G. Seidel, West Virginia University statistician, provided statistical guidance. We thank J. Apodaca for providing comments that improved the quality of this manuscript. We are grateful to Animal and Plant Health Inspection Service, Bureau of Land Management, National Marine Fisheries Service, National Park Service, U.S. Fish and Wildlife Service, U.S. Forest Service, and U.S. Geological Survey for contributions to the J.M.W. position that helped in the development of this manuscript. The study was conducted in compliance with West Virginia University Animal Care and Use guidelines (IACUC protocol #02-0502). Data S1 (*Supplemental Material*) provides data used in analyses.

Any use of trade, product, website, or firm names in this publication is for descriptive purposes only and does not imply endorsement by the U.S. Government.

References

Acton PM, Fox JF, Campbell JE, Jones AL, 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.

Ash, AN. 1997. Disappearance and return of plethodontid salamanders to clearcut plots in the southern Blue Ridge Mountains. Conservation Biology 11:983–989.

Bailey LL, Simons TR, Pollock KH. 2004. Estimating detection probability parameters for *Plethodon* salamanders using the robust capture–recapture design. Journal of Wildlife Management 68:1–13.

Beers TW, Dress PE, Wensel LC. 1966. Aspect transformation in site productivity research. Journal of Forestry 64:691–692.

Bernhardt ES, Palmer MA. 2011. The environmental costs of mountaintop mining valley fill operations for aquatic ecosystems of the central Appalachians. Annals of the New York Academy of Science 1223:39–57.

Blaustein AR, Wake DB, Sousa WP. 1994. Amphibian declines: judging stability, persistence, and susceptibility of populations to local and global extinctions. Conservation Biology 8:60–71.

Borcard D, Gillet F, Legendre P. 2011. Numerical ecology with R. New York: Springer.

Brady JK. 2015. Salamander diversity of reforested abandoned coal surface mines in the Appalachian region, USA. Restoration Ecology 24:398–405.

Brenner, FJ, Brenner EK, Brenner PE. 1992. Analysis of drift fence arrays as a census method for vertebrate communities on a proposed mine site. Journal of the Pennsylvania Academy of Science 65:117–122.

Burnham, KP, Anderson DR, Huyvaert KP. 2011. AIC model selection and multimodel inference in behavioral ecology: some background, observations, and comparisons. Behavioral Ecology and Sociobiology 65:23–35.

Bury RB. 1983. Differences in amphibian populations in logged and old growth redwood forest. Northwest Science 57:167–178.

Caughley G, Gunn A. 1995. Conservation biology in theory and practice. Cambridge, Massachusetts: Blackwell Science.

Chamblin HD. 2002. Small mammal communities on a reclaimed mountaintop mine valley fill landscape in southern West Virginia. Master's thesis. Morgantown: West Virginia University (see *Supplemental Material*, Reference S1, http://dx.doi.org/10.3996/102016-JFWM-079.S2).

Connette GM, Semlitsch RD. 2013. Life history as a predictor of salamander recovery rate from timber harvest in southern Appalachian forests, U.S.A. Conservation Biology 27:1399–1409.

Corn SP 1994. Straight line drift fences and pitfall traps. Pages 109–117 in Heyer WK, Donnelly MA, McDarmid RW, Hayek LC, Foster MS, editors. Measuring and monitoring biological diversity: standard methods for amphibians. Washington, D.C.: Smithsonian Institute Press.

Crawford JA, Semlitsch RD. 2008. Post-disturbance effects of even-aged timber harvest on stream salamanders in southern Appalachian forests. Animal Conservation 11:369–376.

deMaynadier PG, Hunter ML. 1995. The relationship between forest management and amphibian ecology: a review of the North American literature. Environmental Review 3:230–261.

Dodd CK. 2013. Frogs of the United States and Canada. Volumes 1 and 2. Baltimore, Maryland: Johns Hopkins University Press.

Duguay JP, Wood PB. 2002. Salamander abundance in regenerating forest stands on the Monongahela National Forest, West Virginia. Forest Science 48:331–335.

Enge KM. 1997. Use of silt fencing and funnel traps for drift fences. Herpetological Review 28:30–31.

Enge KM, Marion WR. 1986. Effects of clearcutting and site preparation on herpetofauna of a north Florida flatwoods. Forest Ecology and Management 14:177–192.

Ernst CH, Lovich JE, Barbour RW. 1994. Turtles of the United States and Canada. Washington, D.C.: Smithsonian Institution Press.

Fiske IJ, Chandler RB. 2011. Unmarked: an R package for fitting hierarchical models of wildlife occurrence and abundance. Journal of Statistical Software 43:1–23.

Ford WM, Chapman BR, Menzel MA, Odom RH. 2002. Stand age and habitat influences on salamanders in Appalachian cove hardwood forests. Forest Ecology and Management 155:131–141.

Grant EHC et al. 2016. Quantitative evidence for the effects of multiple drivers on continental-scale amphibian declines. Scientific Reports 6, Article Number 25625 (see *Supplemental Material*, Reference S2, http://dx.doi.org/10.3996/102016-JFWM-079.S3); also available: http://www.nature.com/articles/srep25625 (February 2017).

Green NB, Pauley TK. 1987. Amphibians & reptiles in West Virginia. Pittsburgh, Pennsylvania: University of Pittsburgh Press.

Hobbs RJ, et al. 2005. Novel ecosystems: theoretical and management aspects of the new ecological world order. Global Ecology and Biogeography 15:1–7.

James FC, Shugart HH. 1970. A quantitative method of habitat description. Audubon Field Notes 24:727–736.

Kéry M, Royle JA. 2016. Applied hierarchical modeling in ecology: analysis of distribution, abundance and species richness in R and BUGS. Waltham, Massachusetts: Academic Press.

Lacki MJ, Hummer JW, Fitzgerald JL. 2005. Populations patterns of copperbelly water snakes (*Nerodia erythrogaster neglecta*) in a riparian corridor impacted by mining and reclamation. American Midland Naturalist 153:357–369.

Lannoo M. 2005. Amphibian declines: the conservation status of United States species. Berkeley, California: University of California Press.

Lepš J, Šmilauer P. 2003. Multivariate analysis of ecological data using CANOCO. New York: Cambridge University Press (see *Supplemental Material*, Reference S3, http://dx.doi.org/10.3996/102016-JFWM-079.S4).

Lima AT, Mitchell K, O'Connell DW, Verhoeven J, Cappellen PV. 2016. The legacy of surface mining: remediation, restoration, reclamation and rehabilitation. Environmental Science & Policy 66:227–233.

Martin TE, Paine C, Conway CJ, Hockachka WM, Allen P, Jenkins W. 1997. BBIRD Field Protocol. Missoula, Montana: University of Montana (see *Supplemental Material*, Reference S4, http://dx.doi.org/10.3996/102016-JFWM-079.S5); also available: http://www.umt.edu/bbird/docs/BBIRDPROT.pdf (February 2017).

Mazerolle MJ. 2016. Package 'AICcmodavg': model selection and multimodel inference based on (Q)AIC(c). R package version 2.0-4 (see *Supplemental Material*, Reference S5, http://dx.doi.org/10.3996/102016-JFWM-079.S6); also available: https://cran.r-project.org/web/packages/AICcmodavg/ (November 2016).

McCune B, Grace JB. 2002. Analysis of ecological communities. Gleneden Beach, Oregon: MjM Software Design.

NatureServe. 2015. National and subnational conservation status definitions (see *Supplemental Material*, Reference S6, http://dx.doi.org/10.3996/102016-JFWM-079.S7); also available: http://explorer.natureserve.org/nsranks.htm (February 2017).

Oliver CD, Larson BC. 1996. Forest stand dynamics. Update edition. New York: John Wiley & Sons, Inc.

Pais RC, Bonney SA, McComb WC. 1988. Herpetofaunal species richness and habitat associations in an eastern Kentucky forest. Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies 42:448–455.

Petranka JW. 1998. Salamanders of the United States and Canada. Washington, D.C.: Smithsonian Institution Press.

Petranka JW, Eldridge ME, Haley KE. 1993. Effects of timber harvesting on southern Appalachian salamanders. Conservation Biology 7:363–370.

Pike, DA, Webb, JK, Shine, R. 2011. Removing forest canopy cover restores a reptile assemblage. Ecological Applications 21: 274–280.

Pough FH, Andrews RM, Cadle JE, Crump ML, Savitzky AH, Wells KD. 2001. Herpetology. 2nd edition. Upper Saddle River, New Jersey: Prentice Hall.

Pough FH, Smith EM, Rhodes DH, Collazo A. 1987. The abundance of salamanders in forest stands with different histories of disturbance. Forest Ecology and Management 20:1–9.

Robel RJ, Briggs JN, Dayton AD, Hulbert LC. 1970. Relationships between visual obstruction measurements and weight of grassland vegetation. Journal of Range Management 23:295–297.

Ross B, Fredericksen T, Ross E, Hoffman W, Morrison ML, Beyea J, Lester MB, Johnson BN, Fredericksen NJ. 2000. Relative abundance and species richness of herpetofauna in forest stands in Pennsylvania. Forest Science 46:139–146.

Royle JA. 2004. *N*-mixture models for estimating population size from spatially replicated counts. Biometrics 60:108–115.

SAS Institute. 2000. The SAS system for windows: statistical analysis system 8.1. Cary, North Carolina: SAS Institute.

Smith HC. 1977. Height of the tallest saplings in 10-year-old Appalachian hardwood clearcuts. Upper Darby, Pennsylvania: U.S. Forest Service. Research Paper N3-381 (see *Supplemental Material*, Reference S7, http://dx.doi.org/10.3996/102016-JFWM-079.S8); also available: http://www.fs.fed.us/ne/newtown_square/publications/research_papers/pdfs/scanned/ne_rp381p.pdf (February 2017).

Stebbins RC, Cohen NW. 1995. A natural history of amphibians. Princeton, New Jersey: Princeton University Press.

Taub FB. 1961. The distribution of the red-backed salamander (*Plethodon c. cinereus*) within the soil. Ecology 42:681–698.

Thomas KA, Sencindiver JC, Skousen JG, Gorman JM. 2000. Soil horizon development on a mountaintop surface mine in southern West Virginia. Green Lands. Summer:41–51.

Townsend PA, Helmers DP, Kingdon CC, McNeil BE, DeBeurs KM, Eshleman KN. 2009. Changes in the extent of surface mining and reclamation in the central Appalachians detected using a 1976–2006 Landsat time series. Remote Sensing of Environment 113:62–72.

Triska MD, Craig MD, Stokes VL, Pech RP, Hobbs RJ. 2016. The relative influence of in situ and neighborhood factors on reptile recolonization in post-mining restoration sites. Restoration Ecology 24: 517–527.

Vogt RC, Hine RL. 1982. Evaluation of techniques for assessment of amphibian and reptile populations in Wisconsin. Pages 201–217 in Scott NJ, editor. Herpetological communities: a symposium of the Society for the Study of Amphibians and Reptiles and the Herpetologists' League, August 1977. U.S. Fish and Wildlife Service Resource Report No. 13 (see *Supplemental Material*, Reference S8, http://dx.doi.org/10.3996/102016-JFWM-079.S9); also available: https://babel.hathitrust.org/cgi/pt?id=uc1.31822010336709;view=1up;seq=211 (February 2017).

Waldick R. 1997. Effects of forestry practices on amphibian populations in eastern North America. Pages 191–205 in Green, DM, editor. Amphibians in decline: Canadian studies of a global problem (Herpetological Conservation, number 1). St. Louis, Missouri: Society for the Study of Amphibians and Reptiles.

Weakland CA, Wood PB. 2005. Cerulean warbler (*Dendroica cerulea*) microhabitat and landscape-level habitat characteristics in southern West Virginia. Auk 122:497–508.

West Virginia Division of Natural Resources. 2015. 2015 West Virginia State Wildlife Action Plan. South Charleston, West Virginia: West Virginia Division of Natural Resources (see *Supplemental Material*, Reference S9, http://dx.doi.org/10.3996/102016-JFWM-079.S10); also available: http://www.wvdnr.gov/DRAFT%202015%20WV%20State%20Wildlife%20Action%20Plan%20R1.pdf (February 2017).

Wickham JD, Ritters KH, Wade TG, Coan M, Homer C. 2007. The effect of Appalachian mountaintop mining on interior forest. Landscape Ecology 22:179–187.

Williams JM. 2003. Impacts on terrestrial and streamside herpetofauna by mountaintop removal mining in southern West Virginia. Doctoral dissertation. Morgantown, West Virginia: West Virginia University (see *Supplemental Material*, Reference S10, http://dx.doi.org/10.3996/102016-JFWM-079.S11).

Wood PB, Ammer FK. 2015. Grasshopper sparrow reproductive success and habitat use on reclaimed surface mines varies by age of reclamation. Wildlife Society Bulletin 39:512–520.

Wood PB, Bosworth S, Dettmers R. 2006. Cerulean warbler abundance and occurrence relative to large-scale edge and habitat characteristics. Condor 108:154–165.

Wood PB, Williams JM. 2013. Terrestrial salamander abundance on reclaimed mountaintop removal mines. Wildlife Society Bulletin 37:815–823.

Zar JH. 1999. Biostatistical analysis. 4th edition. Upper Saddle River, New Jersey: Prentice Hall.

Zipper CE, Burger JA, Skousen JG, Angel PN, Barton CD, Davis V, Franklin JA. 2011. Restoring forests and associated ecosystem services on Appalachian coal surface mines. Environmental Management 47:751–765.