# Chapter Two Stream-breeding salamander occupancy across the Mid-Atlantic region of the United States

## INTRODUCTION

Understanding the spatial ecology of a species is an integral part of conservation, especially for species with ranges spanning large environmental gradients or experiencing environmental change. Observing species occurrence across heterogeneous landscapes allows us to uncover their associations with habitats, disturbances, and climates. Recent advances in survey design and statistics allow for more accurate estimates of species ranges and the drivers of species distributions. These advances include accounting for imperfect detection of cryptic species to reduce bias and separate ecological patterns from observational processes. Understanding the spatial relationship of plant and animal species is especially important for species that might be declining and are challenging to observe consistently across space and time.

Amphibians are generally cryptic and are experiencing widespread declines and extinctions. Both rare and historically common amphibian species have declined across the United States including in protected conservation areas (Adams et al. 2013). Many factors threaten the abundance and distribution of amphibians, including habitat destruction, disease, pollution, invasive species, climate change, urbanization, and pesticides (e.g. Fedorenkova et al. 2012, Rizzo et al. 2016, Sasaki et al. 2016). Stream-breeding salamanders of the family Plethodontidae are particularly vulnerable to these disturbances due to their complex life cycle, long life span, position in food webs, and physiology. Salamanders can have long lifespans in comparison to other aquatic organisms (Petranka 1998); however, living for such a long time heightens the risk of experiencing a disturbance event. Adults do seem less sensitive than larval stages, so responses can vary across life stage (Kucken et al. 1994). Recognizing factors that affect these integral yet sensitive species should be a priority when managing these stream systems that are continually being disturbed.

Habitat degradation, such as fragmentation and land-use change, is a major threat to stream biota. Fragmentation can prevent movement within a metapopulation and alter assemblage structures. Fragmentation of stream habitat can come in a variety of forms, including road crossings, culverts, and waterfalls. Stream salamander populations are typically negatively affected by habitat fragmentation; and populations often have difficulty returning to their original location if displaced, with the disturbance reducing permeability back into the environment (Cecala et al. 2014). Road crossings have been shown to reduce salamander densities next to the crossing; but some tolerant species (e.g. the northern two-lined salamander, *Eurycea bislineata*) flourished in the abatement of intolerant species. The authors of this study concluded that culverts that are often associated with road crossings over streams probably acted as an impassable barrier due to the pipe outlets hanging in mid-air, separating upstream sections of the stream from downstream (Ward 2005). Resetarits (1997) found that a natural barrier, a waterfall, prevented predatory fish from moving into the waters above; as such, stream salamander assemblage structure was altered downstream in the presence of fish and activity, growth, and survival were all reduced in comparison to that of upstream sections.

Land-use changes are typically characterized by urban sprawl and areas of industrialization and agriculture to support urban areas. Stream ecosystems are particularly sensitive to disturbance because they are dependent on what occurs within the entire watershed, not just the local ecosystem. Land-use, particularly within riparian zones (within a 500 meter radius), may alter multiple habitat features that are important to stream salamanders, such as canopy cover, canopy height, and leaf litter, and thus alter the assemblage structure of salamanders (Surasinghe and Baldwin 2015). Chronic land-use changes have been shown to be associated with long-term, substantial salamander population declines, with one study finding a 32% to 44% decrease in southern two-lined salamander (*Eurycea cirrigera*) populations and a 21% to 30% decrease in northern dusky salamander (*Desmognathus fuscus*) populations due to widespread conversion of forested areas to urban land over the course of three decades (Price et al. 2006).

Climate also serves as a driver of salamander population distributions. Amphibians respire cutaneously (i.e. respire through the skin), which makes them highly dependent on the availability of moisture. Precipitation is a driver of individual growth rates, ground surface activity, and population dynamics of stream salamanders (Bendik and Gluesenkamp 2013, Caruso et al. 2014, Connette et al. 2015). Milanovich et al. (2006) also found that clutch size is related to annual precipitation, implying that future population size and assemblage structure is associated with precipitation. Temperature is another important environmental factor to stream salamanders because they are ectotherms and are dependent on their environment to regulate their metabolic rate (Fitzpatrick 1973). Growth rate, body size at metamorphosis, and length of larval period are also affected by temperature (Beachy 1995), with the latter relationship showing the potential contribution of temperature to future population dynamics.

Understanding relationships between these landscape-scale environmental covariates should be a priority when managing stream systems, due to the important role that plethodontid salamanders serve in local ecosystems. The main objectives of this study were to assess the effects of landscape-scale environmental variables on the occupancy of stream-breeding salamanders in the mid-Atlantic region of the United States across space and time.

## METHODS

### Salamander Data

We collected salamander occupancy data from four different areas within the mid-Atlantic (Western Maryland, Shenandoah National Park, Canaan Valley National Wildlife Refuge, and the National Capital Region of Maryland and Virginia; Figure 12). Stream salamander species observed were the northern dusky salamander (*Desmognathus fuscus*), Allegheny Mountain dusky salamander (*Desmognathus ochrophaeus*), seal salamander (*Desmognathus monticola*), northern two-lined salamander (*Eurycea bislineata*), three-lined salamander (*Eurycea guttoline*ata), long-tailed salamander (*Eurycea longicauda*), northern spring salamander (*Gyrinphilus porphyriticus*), and northern red salamander (*Pseudotriton ruber ruber*). We describe survey details for each location below. Surveys were conducted between 2001 and 2018 in a total of 218 transects.

#### Western Maryland

We surveyed 36 in-stream transects that were 25 meters in length in nine streams, with 20 meters separating each transect. Starting at the downstream end of each transect and traveling upstream, we flipped rocks the size of cobble (between 6 and 25 cm in diameter; Lowe 2005) within the wetted width and captured and identified any salamanders to species that were seen. Each transect was visited one to five times (mean of 1.8 visits), with only one visit per day. All visits were conducted between 29 May to 10 August 2018 (Further details in Chapter 1).

#### National Capital Region

We surveyed 126 in-stream transects that were 20 meters in length, with 20 meters separating each transect. Starting at the downstream end of each transect we searched for stream salamanders under rocks categorized as cobble within the wetted width of the stream. All salamanders were identified to species and put in a plastic bag beside the stream. We used a multi-pass removal method (Dodd 2010), with one to three passes per transect (a mean of 2.6 passes per transect). The salamanders were placed back in the area where they were found after each pass. All surveys were conducted between 3 May and 7 August from 2005 through 2017, with each transect sampled once per year.

#### Canaan Valley National Wildlife Refuge

We surveyed nine in-stream transects that were 20 meters in length, with 20 meters separating each transect. Transects were sampled once a year, from 2001 through 2006 between 11 July and 1 August. Starting at the downstream end of each transect, we searched for stream salamanders under rocks categorized as cobble within the wetted width of the stream. All salamanders were identified to species and put in a plastic bag beside the stream. We used a multi-pass removal method (Dodd 2010), with one to four passes per transect (a mean of 2.5 passes per transect). The salamanders were put back in the area where they were found after each pass.

#### Shenandoah National Park

We surveyed 57 in-stream transects that were 20 meters in length, with 20 meters separating each transect. Transects were sampled once a year, from 01 June 2012 through 25 July 2012. Starting at the downmost point of each transect we searched for stream salamanders under rocks categorized as cobble within the wetted width of the stream. All salamanders were identified to species and put in a plastic bag beside the stream. We used a multi-pass removal method (Dodd 2010), with one to four passes per transect (an average of 3.2 passes per transect). The salamanders were put back in the area where they were found after each pass.

### Landscape-scale Environmental Data Collection

We used environmental and weather data from the Spatial Hydro-Ecological Decision System (SHEDS; ecosheds.org). The SHEDS project gathered land-use and land-cover data from the National Land Cover Dataset (NLCD) 2011 Edition (Jin et al. 2013). Weather data were gathered from Daymet (Thornton et al. 2018), and hydrological information and catchment delineations were derived from the USGS high-resolution National Hydrography Dataset (NHD; [https://www.usgs.gov/ core-science-systems/ngp/national-hydrography/national-hydrography-dataset?qt-science\_ support\_page\_](https://www.usgs.gov/%20core-science-systems/ngp/national-hydrography/national-hydrography-dataset?qt-science_%20support_page_) related\_con=0#qt-science\_support\_page\_related\_con) and corrected for spatial inconsistencies and flow directions. All data for SHEDS were then spatially weighted by hydrological unit code (HUC; <https://ecosheds.org/>#datasets). The SHEDS data products do not yet include 2018 Daymet data, therefore we gathered these data for each transect using the package *daymetr* (Hufkens et al. 2018). We used percent forest cover in the catchment containing the stream transect, mean slope of the catchment, mean annual air temperature (averaged over the previous 30 years), and mean daily precipitation (averaged over the previous 30 years) as covariates affecting occupancy and total precipitation over the previous seven days as a covariate affecting detection. We also allowed occupancy probability to vary randomly by transect.

### Analysis

We developed single-species dynamic occupancy models accounting for imperfect detection to observe the effects of the environmental variables on the occupancy of plethodontid salamanders (Mackenzie 2005, Zipkin et al. 2012). We assumed the populations were closed during the sampling seasons but open during other parts of the year between sampling periods. Occupancy was estimated using the following equations:

(Equation 1)

(Equation 2)

(Equation 3)

(Equation 4)

We modeled the true occupancy state *Z* of site *i* in year *t* as a Bernoulli random variable with a probability of occupancy where = 1 when a species is present at site *i* in year *t*, otherwise zero ((Equation 1). Detection *y* of site *i* during pass *j* in year *t* also followed a Bernoulli distribution where the detection probability *p* at site *i* during pass *j* in year *t* was dependent on the presence of the species (Equation 2). We developed an autologistic model in which we estimated the occupancy state of site *i* in year *t* which was a function of the intercept estimate t site *i* and the estimate *β*1 of the effect of forest cover (forest) at site *i* and the estimate *β*2 of the effect of slope (slope) at site *i* and the estimate *β*3 of the effect of mean air temperature () at site *i*, and the estimate *β*4 of the effect of precipitation (precip) at site *i.* and the estimate *β*5 of the effect of whether or not the species was present in the previous year (Equation 3). Detection was similarly modelled with the probability of detection *p* at site *i* during pass *j* in year *t* estimated by an estimated intercept *α*0, an estimate of the effect *α*1 of the total precipitation from the previous seven days (*precip7*) of site *i* during pass *j*, and an estimate of the effect of the multi-pass removal method *α*2 for each pass *j*, which was not included for the Western Maryland sites that were sampled once a day for multiple visits (Equation 4).

We fit the model using Just Another Gibbs Sampler (JAGS; Plummer 2003) with the *jagsUI* package (Kellner 2019) The priors of the coefficients on the logit of psi and p were normal distributions with a mean of 0 and standard deviation of -1.5 (formulated as the precision in JAGS for computational purposes). The priors on the standard deviation of psi’s intercept were also defined by a half Cauchy distribution with a location of 0 and a scale of 1.3.

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## RESULTS

We observed a total of 2704 salamanders over the course of 17 years; the average number of individuals observed per transect per visit was 3.23 ± 2.59 individuals (Table 9). Average detection across all transects was 0.772 ± 0.023 for the northern dusky salamander; 0.8069 ± 0.0373, the seal salamander; 0.643 ± 0.066, Allegheny Mountain dusky salamander; 0.813 ± 0.015, northern two-lined salamander; 0.584 ± 0.065, three-lined salamander; 0.403 ± 0.135, long-tailed salamander; 0.506 ± 0.044, northern red salamander; and 0.738 ± 0.041, northern spring salamander. The species with the highest estimated occupancy was the northern two-lined salamander, while the species with the lowest estimated occupancy was the three-lined salamander (overall means = 167.36 ± 4.22 sites estimated occupied and 16.71 ± 1.52 sites estimated occupied, respectively; derived from Table 11-11; Figure 13). The results of the model showed little turnover, or when a transect that is occupied becomes unoccupied the next year or vice versa (overall mean = 11.81% ± 0.75%, derived from Table 11-11; Figure 14). Turnover varied by species, with northern red salamanders showing the highest amount of turnover (20.18% ± 0.93%) and seal salamanders the lowest amount of turnover (5.24% ± 0.58%).

Each species showed differing responses to model covariates affecting occupancy (). Forest cover had a significantly positive effect on the seal and two-lined salamander and a negative effect on the long-tailed salamander (Figure 15; Table 12, 7, and 9). Slope had a significantly positive effect on the northern dusky, seal, two-lined, and northern spring salamander and a negative effect on the northern red salamander (Figure 15; Table 11, 5, 7, 10, 11). The mean air temperature had a significantly positive effect on the three-lined salamander and a negative effect on the northern dusky salamander (Figure 15; Table 11 and Table 15). The Allegheny Mountain dusky salamander was not significantly affected by any of the covariates in the model (Figure 15; Table 13). In addition, none of the species showed a significant relationship with any of the covariates affecting detection, including the effect of multi-pass removal sampling and total precipitation from seven days prior to sampling (Figure 16).

## DISCUSSION

Our results indicate that stream salamander populations are relatively stable in the Mid-Atlantic region, with little overall turnover of occupancy and little variance in turnover rates across years. Northern two-lined salamanders were the most prolific species, while the Allegheny Mountain Dusky salamander was the most cryptic species. The results of the model also showed that responses to landscape-scale environmental covariates vary by species. Forest cover had positive effects on seal and two-lined salamanders and a negative effect on long-tailed salamanders. These relationships indicate that the former two species prefer forested streams, thus making them more sensitive to land-use change and urbanization, while the latter species is not as dependent on the presence of forest. Previous studies have shown overall abundances of seal salamanders are negatively affected by reductions in forest cover, particularly related to timber harvest (Crawford and Semlitsch 2008, Moseley et al. 2008). Similarly, two-lined salamanders can be sensitive to loss of forested habitat due to logging (Perkins and Hunter Jr. 2006) because they often migrate from their aquatic habitat into nearby terrestrial ecosystems (Petranka 1998). The negative relationship of forest cover with long-tailed salamanders could be due this species’ ability to move over long distances (Nazdrowicz 2015), which could lessen the overall effect of disturbances in a forested area.

The steepness of the catchment also had positive relationships with most species of salamanders, including the northern dusky, seal, two-lined, and northern spring salamander. The only species that was negatively affected by slope was the northern red salamander. The slope of a stream and associated substrate are known to impact salamander distribution in other regions of the United States (Barr and Babbitt 2002, Lowe et al. 2004, Martin et al. 2012). Low gradient streams encourage accumulation of fine sediments, which fill interstitial spaces and embed larger rocks (Lowe and Bolger 2002). Stream salamanders are generally associated with rocky-bottomed streams (Rocco and Brooks 2000), where they use the interstitial spaces for cover from predators. Sedimentation is also known to have a negative effect on abundance of stream salamanders because it prevents them from seeking such cover (Lowe et al. 2004, Surasinghe and Baldwin 2015). However, salamanders seem to also be negatively affected by rocky-bottomed streams with relatively large substrate and interstitial spaces due to the predators associated with those types of habitat. One study showed that the abundance of the Oklahoma salamander, *Eurycea tynerensis*, was significantly higher in medium-sized substrate (between 12 and 50 millimeters in diameter), avoiding fish in open areas and crayfish in large-sized substrate (Martin et al. 2012). The northern red salamander may be tolerant of streams that are affected by sedimentation and other seemingly less desirable habitat. Larvae and adults typically inhabit leaf packs, which often occur in slow-moving waters of springs and small streams where the leaves are allowed to accumulate (Martof 1975). Northern red salamanders seem to depend more upon these leaf packs to hide from predators than interstitial spaces in rocky-bottomed streams commonly associated with other salamander species.

The mean air temperature had a positive effect on three-lined salamanders and a negative effect on northern dusky salamanders. Previous studies have shown that stream salamanders are affected by water temperature (Barr and Babbitt 2002). We used air temperature as a proxy for water temperature to determine the gross effect of temperature on stream salamanders. The results of the models contradict previous research of temperature preferences and tolerances (Moore 2000, Marvin 2003), suggesting that air temperature may not be a viable proxy for water temperature when observing its effects on stream biota. Also, the extent of this study was at the northernmost extent of the three-lined salamander’s species range (Petranka 1998). This species is adapted to warmer temperatures typical of southern streams, which could explain why they were associated with warmer streams within the Mid-Atlantic region. The northern dusky salamander’s species range is relatively large in comparison to other plethodontid salamanders, stretching from northern Georgia to southeastern Canada (Petranka 1998). The extent of this study was set in the southernmost region of the northern dusky’s species range, which could also show why this species associated with cooler streams within the Mid-Atlantic region.

Other studies have linked salamander abundances and distributions to fine-scale environmental covariates, such as water temperature, water chemistry, and physical features (Barr and Babbitt 2002, Grant et al. 2005, Lowe 2005) and landscape-scale parameters, such as forest cover, connectivity, and impervious surfaces (Lowe and Bolger 2002, Barrett et al. 2010, Surasinghe and Baldwin 2015). This research, along with our own, show how variable stream salamander responses to disturbances can be in different regions or amongst different species. Individuals in differing life stages also display contradicting reactions to environmental perturbances (Lowe 2005), which was something that we did not account for in our models. Including life stage and other landscape-scale environmental covariates could aid in parsing out the underlying trends that affect stream salamander distributions. Future studies should be done on larger scales and in regions with little data to inform managers of local and regional patterns, which seem to lack consistency across the United States.

## LITERATURE CITED

Adams, D. B., K. Burke, B. Hemingway, J. Keay, and M. Yurewicz. 2001. Metal contamination and acid drainage associated with abandoned metal and sulfur mines in the Appalachian Region. Pages 23–32 *in*. U . S. Geological Survey Appalachian Region Integrated Science Workshop Proceedings, Gatlinburg, Tennessee, October 22-26, 2001.

Adams, M. J., D. A. W. Miller, E. Muths, P. S. Corn, E. H. C. Grant, L. L. Bailey, G. M. Fellers, R. N. Fisher, W. J. Sadinski, H. Waddle, and S. C. Walls. 2013. Trends in Amphibian Occupancy in the United States. PLoS ONE 8:6–11.

ArcGIS. 2018. Environmental Systems Research Institute (ESRI).

Bank, M. S., C. S. Loftin, and R. E. Jung. 2005. Mercury bioaccumulation in northern two-lined salamanders from streams in the northeastern United States. Ecotoxicology 14:181–191.

Barr, G. E., and K. J. Babbitt. 2002. Effects of biotic and abiotic factors on the distribution and abundance of larval two-lined salamanders (Eurycea bislineata) across spatial scales. Oecologia 133:176–185.

Barrett, K., B. S. Helms, C. Guyer, and J. E. Schoonover. 2010. Linking process to pattern: Causes of stream-breeding amphibian decline in urbanized watersheds. Biological Conservation 143:1998–2005. Elsevier Ltd.

Bates, D., M. Maechler, B. Bolker, and S. Walker. 2015. Fitting linear mixed-effects models using lme4. Journal of Statistical Software 67:1–48.

Beachy, C. K. 1995. Effects of larval growth history on metamorphosis in a stream-dwelling salamander (Desmognathus ochrophaeus). Journal of Herpetology 29:375–382.

Bendik, N. F., and A. G. Gluesenkamp. 2013. Body length shrinkage in an endangered amphibian is associated with drought. Journal of Zoology 290:35–41.

Bernhardt, E. S., M. A. Palmer, J. D. Allan, G. Alexander, K. Barnas, S. Brooks, J. Carr, S. Clayton, C. Dahm, J. Follstad-Shah, D. Galat, S. Gloss, P. Goodwin, D. Hart, B. Hassett, R. Jenkinson, S. Katz, G. M. Kondolf, P. S. Lake, R. Lave, J. L. Meyer, T. K. O’Donnell, L. Pagano, B. Powell, and E. Sudduth. 2005. Synthesizing U.S. river restoration efforts. Ecology 308:636–637.

Brand, A. B., and E. H. C. Grant. 2017. Design tradeoffs in long-term research for stream salamanders. Journal of Wildlife Management 81:1430–1438.

Bruce, R. C. 1989. Life history of the salamander Desmognathus monticola, with a comparison of the larval periods of D. monticola and D. ochrophaeus. Herpetologica 45:144–155.

Caruso, N. M., M. W. Sears, D. C. Adams, and K. R. Lips. 2014. Widespread rapid reductions in body size of adult salamanders in response to climate change. Global Change Biology 20:1751–1759.

Cecala, K. K., W. H. Lowe, and J. C. Maerz. 2014. Riparian disturbance restricts in-stream movement of salamanders. Freshwater Biology 59:2354–2364.

Connette, G. M., J. A. Crawford, and W. E. Peterman. 2015. Climate change and shrinking salamanders: Alternative mechanisms for changes in plethodontid salamander body size. Global Change Biology 21:2834–2843.

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

Day, P. R., and W. M. Forsythe. 1957. Hydrodynamic dispersion of solutes in the soil moisture stream. Soil Science Society of America 21:477–480.

DeMali, H. M., S. E. Trauth, and J. L. Bouldin. 2016. Metals, parasites, and environmental conditions affecting breeding populations of spotted salamanders (Ambystoma maculatum) in Northern Arkansas, USA. Bulletin of Environmental Contamination and Toxicology 96:732–737.

Dodd, C. K. 2010. Amphibian ecology and conservation: A handbook of techniques. Oxford University Press, Oxford, United Kingdom.

Drever, J. L. 1997. The geochemistry of natural waters: Surface and groundwater environments. Third edition. Prentice-Hall, Inc., Upper Saddle River, New Jersey.

Egea-Serrano, A., R. A. Relyea, M. Tejedo, and M. Torralva. 2012. Understanding of the impact of chemicals on amphibians: A meta-analytic review. Ecology and Evolution 2:1382–1397.

Ennen, J. R., J. M. Davenport, and K. F. Alford. 2016. Evidence for asymmetric competition among headwater stream vertebrates. Hydrobiologia 772:207–213.

Equeenuddin, S. M., S. Tripathy, P. K. Sahoo, and M. K. Panigrahi. 2012. Metal behavior in sediment associated with acid mine drainage stream: Role of pH. Journal of Geochemical Exploration 124:230–237.

Fedorenkova, A., J. A. Vonk, H. J. R. Lenders, R. C. M. Creemers, A. M. Breure, and A. J. Hendriks. 2012. Ranking ecological risks of multiple chemical stressors on amphibians. Environmental Toxicology and Chemistry 31:1416–1421.

Fitzpatrick, L. C. 1973. Influence of seasonal temperatures on the energy budget and metabolic rates of the northern two-lined salamander, Eurycea bislineata bislineata. Comparative Biochemistry and Physiology 45:807–818.

Freda, J. 1986. The influence of acidic pond water on amphibians: a review. Water, Air, and Soil Pollution 30:439–450.

Grant, E. H. C., R. E. Jung, and K. C. Rice. 2005. Stream salamander species richness and abundance in relation to environmental factors in Shenandoah National Park, Virginia. The American Midland Naturalist 153:348–356.

Green, L. E., and J. E. Peloquin. 2008. Acute toxicity of acidity in larvae and adults of four stream salamander species (Plethodontidae). Environmental Toxicology and Chemistry2 27:2361–2367.

Hogsden, K. L., and J. S. Harding. 2012. Consequences of acid mine drainage for the structure and function of benthic stream communities: a review. Freshwater Science 31:108–120.

Hufkens, K., D. Basler, T. Milliman, E. K. Melass, and A. D. Richardson. 2018. An integrated phenology modelling framework in R: modelling vegetation phenology with phenor. Methods in Ecology and Evolution 9:1–10.

Jin, S., L. Yang, P. Danielson, C. Homer, J. Fry, and G. Xian. 2013. A comprehensive change detection method for updating the National Land Cover Database to circa 2011. Remote Sensing of Environment 132:159–175.

Jung, R., P. Nanjappa, and E. Grant. 2004. Stream salamander monitoring: Northeast refuges and parks. Northeast Amphibian Research and Monitoring Initiative.

Kellner, K. n.d. Jagsui: a wrapper around “rjags” to streamline “JAGS” analyses. 2019. <https://cran.r-project.org/package=jagsUI>.

Konhauser, K. 2007. Formation of acid mine drainage. Page 220 *in*. Introduction to geomicrobiology. Blackwell Publishing, Malden, Massachusetts.

Kucken, D. J., J. S. Davis, J. W. Petranka, and C. K. Smith. 1994. Anakeesta stream acidification and metal contamination: Effects on a salamander community. Journal of Environmental Quality 23:1311–1317.

LEO EnviroSci Inquiry. 2011. Acid mine drainage in the Mid-Atlantic region (EPA). Lehigh University. <http://www.ei.lehigh.edu/envirosci/enviroissue/amd/links/graphs.html>.

Lowe, W. H. 2005. Factors affecting stage-specific distribution in the stream salamander Gyrinophilus Porphyriticus. Herpetologica 61:135–144.

Lowe, W. H., and D. T. Bolger. 2002. Local and landscape-scale predictors of salamander abundance in New Hampshire headwater streams. Conservation Biology 16:183–193.

Lowe, W. H., K. H. Nislow, and D. T. Bolger. 2004. Stage-specific and interactive effects of sedimentation and trout on a headwater stream salamander. Ecological Applications 14:164–172.

Mackenzie, D. I. 2005. Was it there? dealing with imperfect detection for species presence/absence data. Australian and New Zealand Journal of Statistics 47:65–74.

Martin, S. D., B. A. Harris, J. R. Collums, and R. M. Bonett. 2012. Life between predators and a small space: substrate selection of an interstitial space-dwelling stream salamander. Journal of Zoology 287:205–214.

Martof, B. S. 1975. Pseudotriton ruber (Latreille): Red salamander. Pages 1–3 *in*. Catalogue of American amphibians and reptiles.

Marvin, G. A. 2003. Effects of acute temperature and thermal acclimation on aquatic and terrestrial locomotor performance of the three-lined salamander, Eurycea guttolineata. Journal of Thermal Biology 28:251–259.

Mazerolle, M. J. 2019. AICcmodavg: Model selection and multimodel inference based on (Q)AIC(c). R package.

McCormick, P. G. 1972. The determination of dissolved oxygen by the Winkler method: A student laboratory experiment. Journal of Chemical Education 49:839.

Milanovich, J. R., S. E. Trauth, D. A. Saugey, and R. R. Jordan. 2006. Fecundity, reproductive ecology, and influence of precipitation on clutch size in the western slimy salamander (Plethodon albagula). Herpetologica 62:292–301.

Monahan, R., and M. Stover. 2018. Maryland’s draft 2018 integrated report of surface water quality. Baltimore, Maryland.

Moore, C. M. 2000. Temperature-mediated characteristics of the dusky salamander (Desmognathus fuscus) of southern Appalachia. Emporia State University.

Moseley, K. R., W. Mark Ford, J. W. Edwards, and T. M. Schuler. 2008. Long-term partial cutting impacts on Desmognathus salamander abundance in West Virginia headwater streams. Forest Ecology and Management 254:300–307.

Nazdrowicz, N. H. 2015. Ecology of the eastern long-tailed salamander (Eurycea longicauda longicauda) associated with springhouses. University of Delaware.

Perkins, D. W., and M. L. Hunter Jr. 2006. Effects of riparian timber management on amphibians in Maine. Journal of Wildlife Management 70:657–670.

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

Petranka, J. W. 1998. Salamanders of the United States and Canada. Smithsonian Institution Press, Washington D. C., USA.

Plummer, M. 2003. JAGS: A program for analysis of Bayesian graphical models using Gibbs sampling.

Price, S. J., M. E. Dorcas, A. L. Gallant, R. W. Klaver, and J. D. Willson. 2006. Three decades of urbanization: Estimating the impact of land-cover change on stream salamander populations. Biological Conservation 133:436–441.

R Core Team. 2019. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Resetarits, W. J. 1997. Differences in an ensemble of streamside salamanders (Plethodontidae) above and below a barrier to brook trout. Amphibia-Reptilia 18:15–25.

Resources, M. D. of N. 2018. Brook trout population restored in western Maryland watershed. 4 December 2018.

Rizzo, A. A., R. L. Raesly, and R. R. Hilderbrand. 2016. Stream salamander responses to varying degrees of urbanization within Maryland’s piedmont physiographic province. Urban Ecosystems 19:397–413.

Rocco, G. L., and R. P. Brooks. 2000. Abundance and distribution of a stream plethodontid salamander assemblage in 14 ecologically dissimilar watersheds in the Pennsylvania Central Appalachians. Prepared for U.S. Environmental Protection Agency, Region III.

Rowe, C. L., and J. Freda. 2000. Effects of acidification on amphibians at multiple levels of biological organization. Pages 545–571 *in*. Ecotoxicology of Amphibians and Reptiles. SETAC Press, Pensacola.

Sasaki, K., D. Lesbarrères, C. T. Beaulieu, G. Watson, and J. Litzgus. 2016. Effects of a mining-altered environment on individual fitness of amphibians and reptiles. Ecosphere 7:1–14.

Southerland, M., R. Jung, D. Baxter, I. Chellman, G. Mercurio, and J. Vølstad. 2004. Stream salamanders as indicators of stream quality in Maryland, USA. Applied Herpetology 2:23–46.

Stranko, S., S. Smith, L. Erb, and D. Limpert. 2010. A key to the amphibians and reptiles of Maryland. Maryland Department of Natural Resources, Annapolis, Maryland.

Surasinghe, T. D., and R. F. Baldwin. 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.

Thornton, M. M., P. E. Thornton, Y. Wei, B. W. Mayer, R. B. Cook, and R. S. Vose. 2018. Station-level inputs and model predicted values for North America, Version 3. Oak Ridge, Tennessee, USA.

Vatnick, I., J. Andrews, M. Colombo, H. Madhoun, M. Rameswaran, and M. A. Brodkin. 2006. Acid exposure is an immune disruptor in adult Rana pipiens. Environmental Toxicology and Chemistry 25:199–202.

Ward, R. L. 2005. The effects of roads and culverts on stream and stream-side salamander communities in eastern West Virginia. West Virginia University.

Ward, R. L., J. T. Anderson, and J. T. Petty. 2008. Effects of road crossings on stream and streamside salamanders. Journal of Wildlife Management 72:760–771.

Waters, A. S., and J. G. Webster-Brown. 2016. Is dilution a solution to aluminum toxicity in an acid mine drainage affected stream o nthe Stockton Plateau, New Zealand? Mine Water and the Environment 35:235–242.

Wells, K. D. 2007. The ecology and behavior of amphibians. The University of Chicago Press, Chicago.

Williams, K. M., and A. M. Turner. 2015. Acid mine drainage and stream recovery: Effects of restoration on water quality, macroinvertebrates, and fish. Knowledge and Management of Aquatic Ecosystems 1–12.

Woodley, S. K., P. Freeman, and L. F. Ricciardella. 2014. Environmental acidification is not associated with altered plasma corticosterone levels in the stream-side salamander, Desmognathus ochrophaeus. General and Comparative Endocrinology 201:8–15.

Zipkin, E. F., E. H. Campbell Grant, and W. F. Fagan. 2012. Evaluating the predictive abilities of community occupancy models using AUC while accounting for imperfect detection. Ecological Applications 22:1962–1972.