



Evaluation of created wetlands as amphibian habitat on a reforested surface mine

Michaela Lambert^{a,b}, Andrea N. Drayer^{a,*}, Wendy Leuenberger^{a,c}, Steven J. Price^a, Christopher Barton^a

^a University of Kentucky, Department of Forestry and Natural Resources, 730 Rose Street, Lexington, KY 40546, United States of America

^b Kentucky Division of Water, Watershed Management Branch, 300 Sower Blvd, Frankfort, KY 40601, United States of America

^c Michigan State University, Department of Integrative Biology, Ecology, Evolution, and Behavior Program, East Lansing, MI 48824, United States of America

ARTICLE INFO

Keywords:

Reforestation
Created wetland
Occupancy
Abundance
Amphibian
Pond breeding amphibian

ABSTRACT

Wetlands created within disturbed landscapes may be an important key to restoring lost ecosystem functions. Reclaimed mines provide an opportunity to create wetlands and restore natural features within a disturbed landscape while benefiting amphibians, a taxa affected by habitat loss. A large-scale restoration effort within the Monongahela National Forest, West Virginia, USA has resulted in the creation of over 1400 small wetlands. We sampled 39 of these wetlands among four ages (2, 4, 6, and 8 years) to: 1) characterize differences in wetland habitat, 2) estimate amphibian occupancy and abundance, 3) identify wetland characteristics most important for amphibian utilization of wetlands. Mean wetland size, percent canopy cover, percent vegetation cover, distance to the nearest stream, and distance to the nearest forest varied significantly among wetland age classes. Water quality was within normal parameters for natural and created wetlands within our study region. Amphibian occupancy and abundance varied by species, with occupancy predicted by wetland age (Green Frog, *Lithobates clamitans*) and lower percent vegetation cover (Spotted Salamander, *Ambystoma maculatum*, *L. clamitans*) while abundance was predicted by wetland age (*A. maculatum*), higher percent vegetation cover (Eastern Newt, *Notophthalmus viridescens*), larger wetland surface area (*A. maculatum*, *N. viridescens*), and shorter distance to forest (*N. viridescens*). Our results indicate, with adequate site preparation, created wetlands on reforested surface mines provide suitable breeding habitat for pond breeding amphibians.

1. Introduction

To mitigate natural wetland loss, land managers often construct wetlands (Batzler and Sharitz, 2014; Dahl, 2005; Dahl, 2011). Between 1998 and 2004 there was a net gain of 98,014.86 freshwater wetland hectares (ha) in the United States, much of which can be attributed to the creation of over 283,279.95 ha of created open water wetlands, most maintaining deep basins, steep slopes, and limited emergent vegetation (Dahl, 2005; Dahl, 2011). More recently, ecologists have questioned the ability of created wetlands, which are often placed in disturbed landscapes, to provide the same services as natural wetlands (Calhoun et al., 2014; Kudray and Schemm, 2008). Studies have shown that created wetlands placed in degraded landscapes with deep basins, steep slopes and sparse vegetation cover, provide open water systems that often function differently than natural, vegetated wetlands (Minkin and Ladd, 2003; Shulse et al., 2010). Because of this, the ability of created wetlands

to function the same as natural wetlands may be limited if adequate natural features, such as wetland size, hydroperiod, and vegetation cover, are not mimicked (Calhoun et al., 2014; Denton and Richter, 2013; Drayer and Richter, 2016; Drayer et al., 2020; Kross and Richter, 2016; Porej and Hetherington, 2005; Zedler and Kercher, 2005). However, if natural features can be mimicked, created wetlands may provide similar ecological functions to natural wetlands (Brodman et al., 2006; Brown et al., 2012; Calhoun et al., 2014; Dahl, 2005; Drayer and Richter, 2016; Drayer et al., 2020; Kross and Richter, 2016; Rannap et al., 2009).

In an era of global biodiversity loss, amphibians have been recognized as one of the vertebrate taxa most threatened with population declines (Grant et al., 2016; Green et al., 2020; Lannoo, 2005; Wake, 1991). Habitat loss and degradation has been identified as a major cause for declines in amphibians (Gallant et al., 2007; Grant et al., 2016; Semlitsch et al., 2017). Semi-permanent wetlands are particularly important for pond-breeding amphibians, offering a place to breed that

* Corresponding author.

E-mail address: andrea.drayer@uky.edu (A.N. Drayer).

is usually devoid of predators, such as fish (Batzer and Sharitz, 2014; Brown et al., 2012). In addition, the terrestrial habitat surrounding wetlands must also be sufficient to support adult amphibian populations and movement (Birn-Raybuck et al., 2010; Hamer et al., 2016; Marsh et al., 1999; Laan and Verboom, 1990; Price et al., 2018; Semlitsch, 2008; Shulse et al., 2010). Under sufficient conditions, pond-breeding amphibians have the ability to quickly recover loss of local populations because they can produce large clutch sizes and disperse to neighboring wetlands (Gibbons et al., 2006; Hanski and Gilpin, 1991; Semlitsch and Bodie, 1998; Sjögren, 1991; Rannap et al., 2009). In areas where anthropogenic disturbance has led to pond-breeding amphibian declines, there is an opportunity to establish created wetlands with sufficient conditions to allow amphibian populations to rebound.

Surface mining is a common method of coal extraction within the Appalachian region of the U.S. (Bernhardt and Palmer, 2011). During surface mining, the land is stripped of vegetation, the top layers of soil and rock are removed, and the underlying coal seams are mined (Bernhardt and Palmer, 2011). Mines are reclaimed, according to the Surface Mining Control and Reclamation Act of 1977 (Public law 95–87), often as non-native grasslands or forests (Bernhardt and Palmer, 2011; Skousen et al., 2011; Surface Mining Control and Reclamation Act of 1977, 1977). In this process the land is compacted to reduce erosion, restored to its original contour, and re-vegetated with native or non-native plants (Andersen et al., 1989; Plass, 1982; Skousen et al., 2011; Surface Mining Control and Reclamation Act of 1977, 1977). Ecological succession of plant communities on reclaimed mines is often arrested due to soil compaction (Franklin et al., 2012; Sena et al., 2015; Skousen et al., 2011; Zipper et al., 2011). De-compacting the soil and replacing non-native vegetation with native plants on these sites generates an opportunity to create habitats, such as upland wetlands and forests, for wildlife.

Surface mines present an opportunity to create high elevation wetland habitat for pond-breeding amphibians and other wildlife. A few studies have documented amphibian use and reproduction in wetlands located on surface mines. Loughman (2005) found that Eastern Newts (*Notophthalmus viridescens*) were able to breed in natural and artificial wetlands on an abandoned mine in West Virginia, while Spotted Salamanders (*Ambystoma maculatum*) deposited eggs but larvae did not grow or develop. Wood Frogs (*Lithobates sylvaticus*), did not utilize the mine wetlands for reproduction, however several other anuran species did (Loughman, 2005). Loughman (2005) suggested that with proper wetland planning and creation, minelands could be converted to important breeding habitat for amphibians. Lannoo et al. (2009) suggested that mine spoil habitats, including the various wetlands found on them, could be critical habitat to threatened and endangered amphibian species (i.e., Crawfish Frogs (*L. areolatus*)). Similarly, Stiles et al. (2017) documented the colonization of 14 amphibian species to a reclaimed strip mine and suggested that reclaimed, restored, and properly managed post-disturbance landscapes may provide adequate amphibian breeding habitat. It is important to note that all aquatic habitats on minelands are not equivalent as amphibian habitat. Previous studies have demonstrated reduced amphibian species richness, occupancy, abundance, colonization rates, and persistence rates in natural streams affected by coal mining (Hutton et al., 2020; Muncy et al., 2014; Price et al., 2016, 2018). These studies showed elevated specific conductance, sulfate, calcium, magnesium, potassium, and sodium, in streams affected by mining than in unmined forested control streams (Hutton et al., 2020; Muncy et al., 2014; Price et al., 2016). While streams have documented water quality issues on minelands due to overburden weathering, precipitation-fed created wetlands may not have these same water quality issues on mined landscapes.

A large-scale restoration project was initiated on reclaimed mined land in 2010 within the Monongahela National Forest (MNF), West Virginia to bring back the red spruce (*Picea rubens*) ecosystem that once occupied high elevation areas in the state. Prior to disturbance from logging in the early 1900s and surface mining in the 1980s and 1990s,

the red spruce forest covered over 200,000 ha in WV, but less than 20,000 ha remain today (Sullivan, 1993). On the MNF site, a holistic suite of restoration activities have been implemented including soil decompaction, wetland creation, woody debris loading, and planting of native trees and shrubs. To date, over 550,000 trees and shrubs have been planted on nearly 500 ha and over 1400 small wetlands have been constructed on the mine impacted land (French and Barton, 2020). The suitability of these wetlands as habitat for amphibians, however, is not well understood.

To understand restoration outcomes, it is important to investigate how site and landscape-level factors may limit or encourage amphibian use of created wetland habitats on surface mines. Using the wetlands created on the MNF reclaimed mine site, our objectives were to: 1) characterize differences in habitat across wetland age classes, 2) estimate occupancy of five amphibian species *L. sylvaticus*, Green Frog (*L. clamitans*), *N. viridescens*, *A. maculatum*, and Spring Peeper (*Pseudacris crucifer*), 3) estimate abundance of aforementioned species, and 4) identify wetland characteristics most important for amphibian occupancy and abundance of wetlands. Over 1.1 million ha of forest have been disturbed by surface mining in central Appalachia (Bernhardt and Palmer, 2011). By understanding which factors are important to amphibian utilization of created wetlands on high elevation surface mines, there is the possibility for successful reforestation and wetland creation on other Appalachian surface mines.

2. Materials and methods

2.1. Site description

The 16,187 ha Mower Tract is located on Cheat Mountain in the MNF, West Virginia. With an elevation of 4000 ft., the area was historically characterized by Red Spruce - Yellow Birch (*Betula alleghaniensis*) forests, poor soils, a thick layer of peat, and scattered wetlands (Byers et al., 2010). Red spruce forests and their associated systems are of state and global conservation importance and have declined throughout their range in North America (Byers et al., 2010; Walter et al., 2017). The area was logged in the early 1900s and subsequently burned in unnaturally hot wildfires that destroyed the peat and seed-stock. In the 1980s, approximately 809 ha were surface mined for coal. The mined areas were reclaimed by restoring the area to its original contour, compacting the soil, and planting non-native grasses and conifers. Native species were unable to colonize the area due to soil compaction and the pervasiveness of non-native vegetation. The compacted soils also led to slow tree growth and almost nonexistent reproduction. Following reclamation, the land was purchased by the U.S. Forest Service as part of the MNF.

Native reforestation on the Mower Tract began in 2010 using the Forest Reclamation Approach (Burger et al., 2005). To address compaction issues, the area was deep ripped using a bulldozer and a ripping shank in the autumn of each year. Wetland creation was done at the time of ripping using a mid-sized excavator (Komatsu PC 210 or similar) following the methods described by Biebighauser (2011). Wetlands were created opportunistically where clay or wet soils were identified by creating a depression and small berms to allow inundation. Downed trees were left in the ripped area to provide organic material and upland wildlife habitat; logs, downed trees, and larger rocks were placed into the wetlands to create aquatic wildlife habitat. Some of the first wetlands, created in 2010, retained non-native conifer cover and were placed just outside of the ripped area in the adjacent woods, while those created in 2012, 2014 and 2016 were located on the mine benches within the ripped area. There were 135 wetlands created in 2010, 75 in 2012, 279 in 2014, and 318 in 2016. Native red spruce and northern hardwoods were planted in the rip trails, while native wetland shrubs and plants were planted within the wetland and on the wetland edge. Species planted include: Speckled Alder (*Alnus rugosa*), Arrowwood *Viburnum* (*Viburnum recognitum*), Winterberry Holly (*Ilex verticillata*),

Silky Dogwood (*Cornus amomum*), Common Elderberry (*Sambucus canadensis*), Black Chokeberry (*Aronia melanocarpa*), Serviceberry (*Aamelanchier laevis*), Wild Raisin (*Viburnum cassinoides*), Chokecherry (*Prunus virginiana*), Red Osier Dogwood (*Cornus sericea*), and others (French and Barton, 2020). Native wetland plant seed was also spread in and around the wetlands. Not all of these wetlands hold water for all or part of the year. Reforestation and wetland creation efforts on the Mower Tract are ongoing.

2.2. Physical wetland characteristics

Prior to sampling we identified 39 fishless, semi-permanent wetlands, ($n = 10$ for ages 2, 4, and 6; $n = 9$ for age 10). To determine which factors varied among wetland age classes and were important for amphibian occupancy and abundance, we collected site level and environmental variables at each wetland. We counted the number of rocks (diameter ≥ 15 cm) and logs (diameter ≥ 4 cm) and measured water depth manually using a meter stick at all wetlands at the deepest spot at the time of sampling. We measured wetland length and width at the widest points of each wetland basin and multiplied them together to determine wetland surface area (m^2). In July, when vegetation growth was at its peak, we measured percent wetland vegetation and canopy cover. We assessed percent wetland vegetation cover (combined emergent and submergent) visually using a PVC square meter quadrat placed in the middle (2 quadrats) and on the north and south perimeter of the wetland. We averaged all four estimates to calculate overall percent vegetation. We measured canopy cover using a spherical crown densiometer in the center of the wetland (Forestry Suppliers, Jackson, MS, USA). To delineate land cover (forest, grassland), we used National Land Cover Database (NLCD) 2011 3 m resolution land cover data (U.S. Geological Survey, 2011) and we used a stream shapefile supplied by MNF to determine nearest stream locations. We measured distance to nearest stream, wetland, and forest using ArcMap™ 10.5.1 (ESRI, 2013). At each wetland, we took a 50 mL water sample and placed on ice until it could be analyzed in the University of Kentucky Department of Forestry and Natural Resources Hydrology Lab for turbidity (FTU), conductivity ($\mu S\ cm^{-1}$), total organic carbon ($mg\ L^{-1}$), phosphate ($PO_4\ mg\ L^{-1}$), pH (H+), alkalinity ($HCO_3\ mg\ L^{-1}$), chloride ($Cl\ mg\ L^{-1}$), sulfate ($SO_4\ mg\ L^{-1}$), nitrate ($NO_3\ mg\ L^{-1}$), ammonium ($NH_4\ mg\ L^{-1}$), calcium ($Ca\ mg\ L^{-1}$), magnesium ($Mg\ mg\ L^{-1}$), potassium ($K\ mg\ L^{-1}$), sodium ($Na\ mg\ L^{-1}$), manganese ($Mn\ mg\ L^{-1}$), and iron ($Fe\ mg\ L^{-1}$). All sampling, preservation, and analytic protocols followed those outlined in Greenberg et al. (1992).

2.3. Amphibian sampling

We conducted amphibian count surveys five times at each wetland between May and July 2018 using 40 by 23 cm D-frame dipnets (Ed Cumings, Inc.). The number of net sweeps was determined based on wetland surface area prior to sampling, with one sweep per every two m^2 surface area with 5–52 sweeps per wetland (14 ± 1.7). Each dipnet sweep consisted of placing the dipnet on the bottom of the wetland and dragging it approximately 1 m. We sampled all cover types evenly (emergent vegetation, log piles, open water, etc.). We identified and counted adult and larval amphibians on site; however, in a few instances, we confirmed species identification in the lab.

2.4. Data analyses

2.4.1. Physical wetland characteristics

We used an analysis of variance (ANOVA) in SPSS 24 (IBM SPSS Statistics) to look for differences among age classes in all site covariates collected or calculated on a single occasion, including: wetland size (surface area m^2), change in wetland water depth (cm), number of logs, number of rocks, percent canopy cover, percent vegetation cover in

wetland, and distance to nearest stream, wetland, and forest. Additionally, we conducted an ANOVA using water quality data from the second sample. This sample was used because it was the most complete sample (no dry wetlands). We further analyzed significant differences detected by ANOVA's using Tukey's pairwise comparison (Tukey, 1949).

2.4.2. Occupancy and abundance analyses

We analyzed capture data using single season occupancy and N -mixture models using the unmarked package in R (MacKenzie et al., 2002; Royle, 2004; Fiske and Chandler, 2011; R Core Team, 2020). These models account for imperfect detection and calculate occupancy or abundance. The occupancy models fit detection and occupancy with binomial distributions, while N -mixture models fit detection using a binomial distribution and abundance using a Poisson distribution. We estimated occupancy for the five species with adequate data: *A. maculatum*, *N. viridescens*, *L. clamitans*, *L. sylvaticus*, and *P. crucifer*, and used N -mixture models for *A. maculatum*, *N. viridescens*, *L. clamitans*, *L. sylvaticus*. We included covariates of day of year and day of year² on detection for both types of models. We were interested in percent vegetation cover, distance to forest, wetland size, and wetland age as covariates for occupancy and abundance. Wetland age was categorical (age 2, 4, 6, or 8), while all other covariates were continuous. We checked for correlations among continuous variables and did not include variables with correlations >0.7 (Dormann et al., 2013). We checked for collinearity among all covariates and removed covariates if variance inflation factors were >5 (Zuur et al., 2009). No variables were correlated or collinear.

We fit a global model with all terms as additive for each species and response. We conducted goodness-of-fit tests on these models using chi-squared tests with 1000 parametric bootstrapped samples (Burnham and Anderson, 2002; Fiske and Chandler, 2011; Mazerolle, 2016). For the occupancy models, we checked the chi-squared test statistic for adequate fit. For N -mixture models, we looked at the \hat{c} values for overdispersion. If \hat{c} was >1 but <4 , we multiplied the variance-covariance matrix by \hat{c} and used QAICc instead of AICc for model selection (Fiske and Chandler, 2011). If \hat{c} was >4 , we refit the global model using the negative binomial and the zero-inflated Poisson distributions and checked if either had \hat{c} values <4 (Burnham and Anderson, 2002). We used a \hat{c} value of 1 if \hat{c} was estimated as <1 (Kéry and Royle, 2016).

We used a secondary candidate set approach to model selection (Morin et al., 2020). We fit all possible subsets of the detection model while holding the occupancy or abundance components constant with all parameters included in the global model. We then changed the detection model back to the global parameterization and fit all possible subsets of the occupancy or abundance model. We retained any models with a $\Delta QAICc/AIC < 10$ for either of these candidate sets. We then ran a final stage with all combinations of the retained detection and occupancy or abundance models. From this final stage, we selected our supported models based on the likelihood of a model ($\exp[-0.5 \times \Delta QAICc/\Delta AIC]$) of ≥ 0.125 and the absence of uninformative parameters (Burnham and Anderson, 2002; Leroux, 2019). We discuss all terms present in the supported model set. If there were multiple supported models in the supported model set for a given species and response, then we used model-averaged predictions to make graphs of marginal effects. To create these graphs and estimates of each variable, we held other continuous variables at their mean value, or six years for wetland age.

3. Results

3.1. Physical wetland characteristics

We found several differences in habitat parameters among wetland age classes. During the sampling period (May–July) two wetlands dried,

both in the 2 year-old age class. Wetland surface area, percent canopy cover, percent vegetation cover, and distance to nearest stream and forest, ranged from 4.5–121.0 m², 0–99%, 2–74%, 378–1337 m², and 0–182 m², respectively. Wetland surface area (F_{3,35} = 2.93, p = 0.05), percent canopy cover (F_{3,35} = 13.98, p < 0.001), percent vegetation cover (F_{3,35} = 8.90, p < 0.001), distance to the nearest stream (F_{3,35} = 104.19, p < 0.001), and distance to the nearest forest (F_{3,35} = 24.63, p < 0.001) varied significantly among wetland age classes. Conversely, mean wetland change in water level depth (F_{3,35} = 1.28, p = 0.30), number of logs (F_{3,35} = 1.16, p = 0.34), number of rocks (F_{3,35} = 1.80, p = 0.17), and distance to the nearest wetland (F_{3,35} = 1.83, p = 0.16) did not vary significantly among wetland age classes. Wetland surface area was highest in 8 year-old (48.54 ± 11.38) and lowest in 2 year-old wetlands (17.25 ± 3.36). Canopy cover was greater in 8 year-old wetlands (49.19 ± 13.61) compared to the other age classes (2, 0.00 ± 0.00; 4, 0.00 ± 0.00; 6, 2.37 ± 1.59). Percent wetland vegetation cover was significantly greater in 4 year-old wetlands (49.40 ± 6.90) when compared with 2 (14.80 ± 2.29) and 6 (16.80 ± 3.54) year-old wetlands, distance to the nearest forest was highest for the 4 year-old (115.52 ± 12.84) and lowest in the 8 year-old wetlands (6.24 ± 4.05), and distance to the nearest stream was significantly greater for the 2 year-old wetlands (1112.94 ± 52.56), as compared to other age classes (Table 1).

Thirteen of the sixteen water quality parameters varied significantly among age classes including: turbidity (F_{3,33} = 10.64, p < 0.001), conductivity (F_{3,33} = 9.16, p < 0.001), total organic carbon (F_{3,33} = 3.75, p = 0.020), phosphate (F_{3,33} = 9.29, p < 0.001), pH (F_{3,33} = 26.62, p < 0.001), alkalinity (F_{3,33} = 7.71, p < 0.001), sulfate (F_{3,33} = 7.87, p < 0.001), nitrate (F_{3,33} = 11.16, p < 0.001), ammonium (F_{3,33} = 8.05, p < 0.001), calcium (F_{3,33} = 5.52, p = 0.003), magnesium (F_{3,33} = 10.18, p < 0.001), potassium (F_{3,33} = 10.59, p < 0.001), and sodium (F_{3,33} = 4.94, p = 0.006). Turbidity, nitrate, and ammonium were greater in the 2 year-old wetlands and sulfate was higher and pH lower in the 8 year-old wetlands than other age classes. Conductivity was highest in 6 year-old wetlands and lowest in 8 year-old wetlands (Table 2).

3.2. Amphibian communities

Over 2200 amphibian captures were recorded during five sampling trips to each of the 39 wetlands from May to July 2018 with 651 amphibians caught in 8 year-old wetlands, 598 in 6 year-old wetlands, 519 in 4 year-old wetlands, and 475 in 2 year-old wetlands. Five species were

Table 1
Physical wetland characteristics ANOVA results.

	Age 2	Age 4	Age 6	Age 8
Wetland surface area (m ²)*	17.25 ± 3.36 ^a	36.52 ± 8.25 ^{ab}	32.29 ± 5.35 ^{ab}	48.54 ± 11.38 ^b
Change in depth (cm)	11.76 ± 1.16 ^a	9.45 ± 1.10 ^a	9.55 ± 0.26 ^a	11.66 ± 1.66 ^a
Number of logs	0.90 ± 0.28 ^a	2.10 ± 0.84 ^a	1.70 ± 0.45 ^a	2.88 ± 1.23 ^a
Number of rocks	2.40 ± 1.02 ^a	0.40 ± 0.31 ^a	0.20 ± 0.20 ^a	1.88 ± 1.30 ^a
Canopy cover (%)*	0.00 ± 0.00 ^a	0.00 ± 0.00 ^a	2.37 ± 1.59 ^a	49.19 ± 13.61 ^b
Vegetation cover (%)*	14.80 ± 2.29 ^a	49.40 ± 0.07 ^b	16.80 ± 3.54 ^a	29.77 ± 7.53 ^{ab}
Distance to stream (m)*	1112.94 ± 52.56 ^b	464.27 ± 17.96 ^a	504.81 ± 2.87 ^a	533.29 ± 22.77 ^a
Distance to wetland (m)	8.80 ± 0.96 ^a	11.52 ± 2.37 ^a	13.41 ± 1.96 ^a	56.27 ± 35.20 ^a
Distance to forest (m)*	68.25 ± 10.26 ^a	115.52 ± 12.84 ^b	66.66 ± 3.90 ^a	6.24 ± 4.05 ^c

Mean (±SE) wetland size (m²), change in depth (cm), number of logs (diameter ≥ 15 cm), number of rocks (diameter ≥ 4 cm), canopy cover (%), vegetation cover (%), distance to nearest stream (m), distance to nearest wetland (m), and distance to forest (m) for each wetland age class. Significant differences are denoted with an asterisk. Differences among groups are denoted using letters.

Table 2
Wetland water chemistry ANOVA results.

	Age 2	Age 4	Age 6	Age 8
Turbidity (FTU)*	25.51 ± 3.77 ^a	10.79 ± 3.58 ^b	6.12 ± 4.92 ^b	4.16 ± 1.88 ^b
Conductivity (µS cm ⁻¹)*	39.68 ± 2.56 ^{ab}	31.27 ± 2.83 ^{ac}	49.47 ± 5.87 ^b	23.22 ± 1.48 ^c
TOC (mg L ⁻¹)*	2.35 ± 0.14 ^a	2.61 ± 0.26 ^{ab}	3.33 ± 0.17 ^b	3.26 ± 0.34 ^{ab}
PO4 (mg L ⁻¹)*	0.46 ± 0.17 ^{ac}	0.19 ± 0.13 ^a	1.06 ± 0.02 ^b	0.87 ± 0.18 ^{bc}
pH (H+)*	4.89 ± 0.08 ^a	4.77 ± 0.17 ^a	4.82 ± 0.13 ^a	3.20 ± 0.20 ^b
Alkalinity (mg L ⁻¹)*	21.12 ± 3.72 ^a	14.52 ± 3.85 ^{ab}	29.52 ± 6.75 ^a	0.00 ± 0.00 ^b
Cl (mg L ⁻¹)	0.36 ± 0.06 ^a	0.29 ± 0.05 ^a	0.46 ± 0.06 ^a	0.29 ± 0.05 ^a
SO4 (mg L ⁻¹)*	0.81 ± 0.22 ^a	0.89 ± 0.06 ^a	0.74 ± 0.07 ^a	2.15 ± 0.43 ^b
NO3 (mg L ⁻¹)*	0.12 ± 0.02 ^a	0.05 ± 0.02 ^b	0.01 ± 0.01 ^b	0.01 ± 0.02 ^b
NH4 (mg L ⁻¹)*	0.23 ± 0.05 ^a	0.10 ± 0.04 ^{ab}	0.01 ± 0.01 ^b	0.03 ± 0.02 ^b
Ca (mg L ⁻¹)*	0.51 ± 0.08 ^{ab}	0.50 ± 0.16 ^a	0.93 ± 0.10 ^b	0.33 ± 0.06 ^a
Mg (mg L ⁻¹)*	1.15 ± 0.22 ^{ab}	0.55 ± 0.13 ^a	2.11 ± 0.41 ^b	0.30 ± 0.12 ^a
K (mg L ⁻¹)*	1.55 ± 0.11 ^a	0.76 ± 0.09 ^{bc}	1.09 ± 0.12 ^{ab}	0.59 ± 0.08 ^c
Na (mg L ⁻¹)*	0.11 ± 0.01 ^a	0.12 ± 0.01 ^a	0.16 ± 0.01 ^{ab}	0.18 ± 0.02 ^b
Mn (mg L ⁻¹)	0.06 ± 0.03 ^a	0.05 ± 0.01 ^a	0.11 ± 0.03 ^a	0.10 ± 0.03 ^a
Fe (mg L ⁻¹)	0.44 ± 0.05 ^a	0.30 ± 0.08 ^a	0.45 ± 0.06 ^a	0.37 ± 0.07 ^a

Mean (±SE) water quality attributes for each age class. Significant differences are denoted with an asterisk. Differences among groups are denoted using letters.

present in all wetland age categories surveyed, (*A. maculatum*, *L. clamitans*, *L. sylvaticus*, *N. viridescens* and *P. crucifer*); while American toad (*Anaxyrus americanus*) were present in two and eight-year old wetlands and two species were exclusive to eight year-old wetlands, *Hemidactylium scutatum* (Four-Toed Salamander) and *Hyla versicolor* (Gray Treefrog). We captured all but one expected species, Eastern Spadefoot (*Scaphiopus holbrookii*), which were likely not seen due to their irregular breeding patterns (Palis, 2005).

3.3. Amphibian occupancy

Day of year and day of year² were included as detection covariates for *A. maculatum*, and *P. crucifer*, while the null model best explained detection of *N. viridescens*, *L. clamitans*, and *L. sylvaticus*. Percent wetland vegetation and wetland age were included as occupancy covariates for both *A. maculatum* and *L. clamitans* (Tables 3 and 4). *Ambystoma maculatum* (occupancy probability = 0.82, 95% CI = 0.41–0.97) and *L. clamitans* (0.83, 0.39–0.98) estimated occupancy were highest at 2% vegetation cover (Fig. 1). Occupancy probability varied with wetland age for both species with *A. maculatum* ranging from 0.68 (0.39–0.88) in two, 0.56 (0.15–0.90) in four, 0.71 (0.33–0.93) in six, and 0.68 (0.40–0.87) in eight year-old wetlands and *L. clamitans* ranging from 0.26 (0.05–0.67) in two, 0.58 (0.21–0.88) in six, 1.00 (0.00–1.00) in four, and 0.29 (0.07–0.68) in eight year-old wetlands (Fig. 2).

3.4. Amphibian abundance

The supported abundance models for *A. maculatum*, *L. clamitans*, and *L. sylvaticus* included day of year and day of year squared as detection covariates, while the null best explained detection in *N. viridescens*. Wetland size was included as an abundance covariate for both

Table 3

Supported models of amphibian occupancy and abundance from created wetlands on the Monongahela National Forest, West Virginia, USA. We used a secondary candidate set approach to model selection. Models from the final stage with a model likelihood of ≥ 0.125 and without uninformative parameters were considered supported models.

Species ¹	Response	Detection	State	Number of parameters	$\Delta\text{QAIC}_c/\text{AIC}^2$	Likelihood
AMAC	Occupancy	Day of year + day of year ²	Vegetation	5	0.00	1.000
		Day of year + day of year ²	Age	7	0.13	0.937
		Day of year + day of year ²	Null	4	3.04	0.219
NVIR	Abundance	Day of year + day of year ²	Size + age	9	0.00	1.000
		Occupancy	Null	2	1.00	0.607
		Abundance	Vegetation + distance to forest + size	6	0.00	1.000
LCLA	Occupancy	Null	Distance to forest + size	5	2.12	0.346
		Null	Vegetation + age	6	0.00	1.000
		Day of year + day of year ²	Age	5	1.75	0.417
LSYL	Abundance	Day of year	Null	6	0.00	1.000
		Day of year	Null	5	3.99	0.136
		Day of year + day of year ²	Null	2	0.00	1.000
PCRU	Occupancy	Day of year + day of year ²	Null	5	0.00	1.000
		Null	Null	4	0.00	1.000

¹ AMAC = *Ambystoma maculatum*, NVIR = *Notophthalmus viridescens*, LCLA = *Lithobates clamitans*; LSYL = *Lithobates sylvaticus*; PCRU = *Pseudacris crucifer*.

² QAIC_c was used for abundance models, AIC was used for occupancy models.

A. maculatum and *N. viridescens*, while *A. maculatum* also included wetland age and *N. viridescens* included percent wetland vegetation and distance to nearest forest. Abundance for *L. clamitans* and *L. sylvaticus* was best explained by the null (Table 3, Table 4). *Ambystoma maculatum* predicted abundance increased with wetland size, peaking with an abundance of 58.5 (95% CI: 21.6–158.3) individuals at 121 m², though we had few wetlands >69 m², where abundance was 12.7 (5.6–28.6) individuals (Fig. 3). At the mean value of wetland size, *A. maculatum* abundance was highest in the two year-old with 10.4 (5.1–20.8) individuals, eight year old 9.2 (4.4–19.0) individuals, six year-old 4.5 (2.0–10.2) individuals, then four year-old 1.0 (0.2–4.3) individual (Fig. 2). *Notophthalmus viridescens* predicted abundance increased with percent vegetation and wetland size, reaching maximum abundances of 5.8 (2.0–14.7) individuals at 74% vegetation cover and 18.3 (7.3–43.9) individuals at 121 m², or 6.1 (3.5–10.5) individuals at 69 m² and decreased with distance to nearest forest, peaking at 6.4 (3.5–11.7) individuals at the forest edge (distance = 0 m) (Fig. 3).

4. Discussion

Our investigation of created wetlands on MNF identified differences in habitat and water quality parameters among age classes, estimated occupancy and abundance of amphibians utilizing the wetlands, and predicted importance of individual species-specific habitat characteristics. The oldest wetlands tended to be larger in surface area, higher in percent canopy and vegetation cover, and had a shorter distance to the nearest forest. While water quality parameters varied by wetland age, all values were within a normal range for created wetlands (Kutka and Bachmann, 1990; Drayer et al., 2020) and natural wetlands (Brodman et al., 2003; Whigham and Jordan, 2003; Byers et al., 2007). Amphibian occupancy and abundance varied by species, with occupancy of some species predicted by wetland age (*L. clamitans*) and lower percent vegetation cover (*A. maculatum*, *L. clamitans*) while abundance of salamanders was predicted by wetland age (*A. maculatum*), higher percent vegetation cover (*N. viridescens*), larger wetland surface area (*A. maculatum*, *N. viridescens*), and shorter distance to forest (*N. viridescens*). Our results show amphibians occupy and are even abundant in created wetlands located in a reclaimed mine landscape. With the correct site preparation, these sites provide suitable habitat for amphibians.

Much of the physical differences seen between wetland age classes are due to differences in wetland creation methodology. The 8 year-old wetlands were built larger and placed outside of the ripped area in the adjacent forest with conifer tree cover retained. This resulted in higher percent canopy cover values in 8 year-old wetlands than wetlands

created in ripped areas that lacked canopy cover. The placement of some wetlands in the 8 year-old age class within the forest rather than in the ripped areas also resulted in little to no distance to the nearest forest compared to wetlands in other age classes. This proximity to the forest and subsequent retained vegetation resulted in highest percent canopy cover and high percent vegetation cover in the 8 year-old wetlands.

Older wetlands have had a longer period of time for amphibians to colonize them compared to younger wetlands, therefore wetland age has the potential to influence amphibian utilization of created wetlands (Bix-Raybuck et al., 2010; Laan and Verboom, 1990; Lehtinen and Galatowitsch, 2001). Laan and Verboom (1990) found wetland age was the best predictor of the number of species in newly created wetlands and documented a strong relationship between species abundance and colonization rates, likely because older wetlands allowed more time for amphibian colonization and therefore exhibited higher abundances. Bix-Raybuck et al. (2010) found that the presence of four anuran species was associated with the age of retention ponds. Spring Peepers and Bullfrogs (*L. catesbeianus*) were found more in the older ponds, versus Cope's Gray Treefrog (*H. chrysocelis*) and Fowler's Toad (*A. fowleri*), which were found more in the newer wetlands (Bix-Raybuck et al., 2010). However, multiple studies have shown that amphibian species can colonize new wetlands sometimes within a few months since creation, indicating that species dispersal capabilities and habitat connectivity are also likely influential (Lehtinen and Galatowitsch, 2001; Vasconcelos and Calhoun, 2006; Rannap et al., 2009). Our results support the idea that wetlands can be colonized quickly, with the youngest wetlands having the second highest number of species captured (6 species) and the highest estimated abundance for *A. maculatum*. Wetland age was a negative predictor of occupancy for *L. clamitans* for two and eight year old wetlands and a positive predictor of abundance for *A. maculatum* in those same wetland age categories. This inverse relationship likely indicates *A. maculatum* predation on *L. clamitans*, as *A. maculatum* larvae are carnivorous and may prey on anuran tadpoles (Resetarits Jr and Wilbur, 1989; Savage and Zamudio, 2005) rather than a true wetland age relationship.

Rapid colonization of aquatic vegetation within created wetland habitats has been documented (Rannap et al., 2020) and many amphibians, including salamanders and hylids, have demonstrated a positive relationship with aquatic vegetation cover (Shulse et al., 2010, 2012). Wassens et al. (2010) found that the occupancy of waterbodies by Growling Grass Frog (*Litoria raniformis*) was strongly linked to vegetation complexity and that vegetation complexity may reduce the impacts of higher predator densities; however, our observations indicated occupancy rates decreasing with increasing percent vegetation cover for two species, *A. maculatum* and *L. clamitans*. Not all vegetation is

Table 4

Beta coefficients, standard errors (SE), and *p*-values for covariates affecting detection and occupancy or abundance of amphibian species from created wetlands on the Monongahela National Forest, West Virginia, USA.

Species ¹	Response	Model	Component	Coefficient	Estimate	SE	P-value					
AMAC	Occupancy	1	Detection: Day of year + day of year ²	Intercept	0.03	0.35	0.932					
				DOY	1.41	0.38	0.000					
				DOY ²	-1.20	0.39	0.002					
				Occupancy: Vegetation	Intercept	0.70	0.48	0.146				
				Vegetation	-0.91	0.46	0.047					
				2	Detection: Day of year + day of year ²	Intercept	0.07	0.35	0.847			
		DOY	1.41	0.38		0.000						
		DOY ²	-1.20	0.39		0.002						
		Occupancy: Age	Intercept	1.16		0.88	0.190					
		Age 4	-2.43	1.19		0.041						
		Age 6	0.67	1.44		0.640						
		Age 8	-0.19	1.22	0.879							
	3	Detection: Day of year + day of year ²	Intercept	0.06	0.35	0.864						
	DOY		1.42	0.38	0.000							
	DOY ²		-1.20	0.39	0.002							
	Abundance		1	Occupancy: Null	Intercept	0.59	0.39	0.133				
					Detection: Day of year + day of year ²	Intercept	-1.47	0.17	0.000			
					DOY	1.50	0.21	0.000				
		DOY ²		-1.56	0.19	0.000						
		Abundance: Size + age		Intercept	2.34	0.21	0.000					
				Size	0.73	0.06	0.000					
Age 4				-2.31	0.45	0.000						
Age 6				-0.84	0.29	0.003						
Age 8				-0.12	0.28	0.656						
Intercept	-0.06		0.18	0.743								
NVIR	Occupancy	1	Detection: Null	Intercept	1.22	0.42	0.004					
	Abundance	1	Detection: Null	Intercept	-0.70	0.16	0.000					
Abundance: Vegetation + distance to forest + size				Intercept	1.06	0.14	0.000					
				Vegetation	0.40	0.09	0.000					
				Distance to forest	-0.61	0.09	0.000					
				Size	0.49	0.06	0.000					
				Intercept	-0.65	0.15	0.000					
		2	Detection: Null	Abundance: Distance to forest + size	Intercept	1.09	0.13	0.000				
Distance to forest		-0.45			0.10	0.000						
Size		0.61			0.06	0.000						
LCLA		Occupancy			1	Detection: Null	Intercept	0.97	0.21	0.000		
							Occupancy: Vegetation + age	Intercept	-1.42	0.88	0.108	
								Vegetation	-1.65	0.91	0.069	
	Age 4		14.82	71.32				0.835				
	Age 6		1.54	1.03				0.132				
	Age 8		0.46	1.09				0.671				
	2		Detection: Null	Occupancy: Age	Intercept	0.97		0.21	0.000			
	Intercept				-0.40	0.65	0.533					
	Age 4				10.89	59.81	0.856					
Age 6	1.26	0.95			0.185							
Age 8	-0.29	0.96			0.764							
Abundance	1	Detection: Day of year + day of year ²			Intercept	-2.22	0.13	0.000				
			DOY	-0.37	0.05	0.000						
			DOY ²	0.21	0.06	0.001						
			Abundance: Null	Intercept	3.00	0.30	0.000					
				2	Detection: Day of year	Intercept	-2.03	0.11	0.000			
						DOY	-0.44	0.05	0.000			
		Intercept				3.00	0.30	0.000				
		LSYL				Occupancy	1	Detection: Null	Intercept	-0.17	0.31	0.584
									Occupancy: Null	Intercept	-1.00	0.38
			Abundance							1	Detection: Day of year + day of year ²	Intercept
		DOY		-0.62	0.07	0.000						
		DOY ²		-0.57	0.08	0.000						
PCRU	Occupancy	1	Abundance: Null	Intercept	-2.79	0.36	0.000					
				Detection: Day of year + day of year ²	Intercept	0.64	0.56	0.257				
					DOY	2.91	0.99	0.003				
					DOY ²	-3.16	0.87	0.000				
					Occupancy: Null	Intercept	-0.60	0.36	0.092			

¹ AMAC = *Ambystoma maculatum*, NVIR = *Notophthalmus viridescens*, LCLA = *Lithobates clamitans*; LSYL = *Lithobates sylvaticus*; PCRU = *Pseudacris crucifer*.

beneficial to amphibians, as wetland plants that produce litter with high C:N ratios, N:P ratios, and soluble phenolics can negatively influence amphibian development, regardless of whether or not the vegetation is native (Cohen et al., 2012). This may explain why our results contrast others findings. However, relationships with vegetation cover may be species specific as we found *N. viridescens* estimated abundance increased as within-pond vegetation increased. Vegetation plays a key

role in *N. viridescens* egg deposition as females lay eggs singly and wrap each egg in submerged leaves of wetland vegetation (Bishop, 1941). This extra protection guards eggs against predation (Orizaola and Brana, 2003). Wetland vegetation within wetlands at MNF is likely performing this protective function leading to increased *N. viridescens* abundances in wetlands with higher vegetation cover.

Wetland surface area is known to be positively associated with

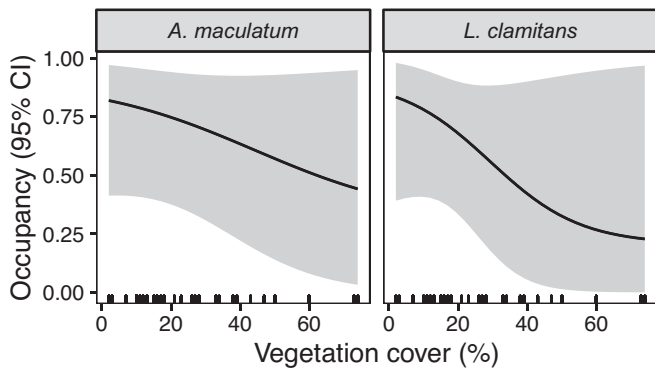


Fig. 1. Marginal effects of vegetation cover on the occupancy of *Ambystoma maculatum* and *Lithobates clamitans*. Original data points are shown in the rug at the bottom of the plot.

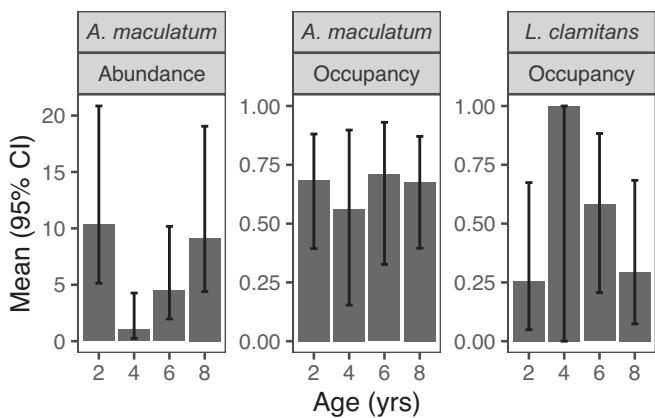


Fig. 2. Marginal effects of age of wetland on the occupancy and abundance of *Ambystoma maculatum* and occupancy of *Lithobates clamitans*.

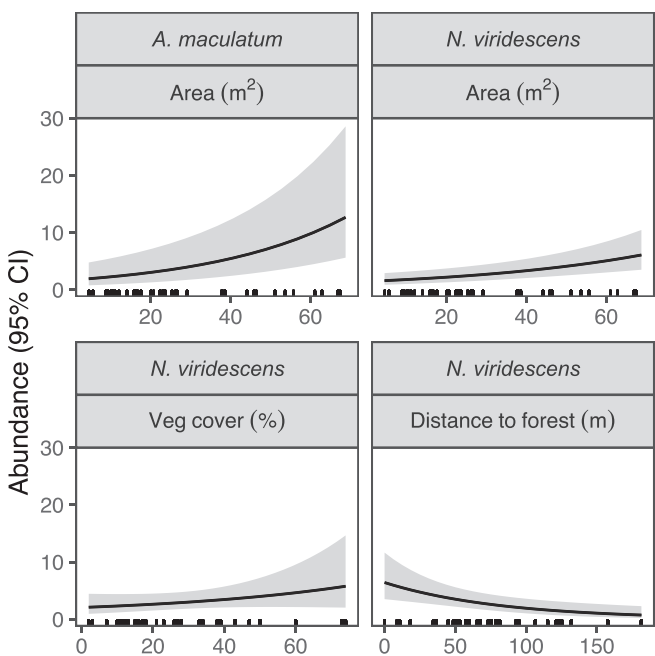


Fig. 3. Marginal effects of wetland characteristics on the abundance of *Ambystoma maculatum* and *Notophthalmus viridescens*. Original data points are shown in the rug at the bottom of the plot.

amphibian species density, richness, and occupancy (Babbitt, 2005; Drayer et al., 2020; Otto et al., 2007; Semlitsch et al., 2015; Werner et al., 2007; Zipkin et al., 2012). Specifically, intermediate wetland sizes (0.1–1.0 ha) with short or intermediate hydroperiods provide habitat for the highest number of amphibian species (Babbitt, 2005; Drayer et al., 2020; Otto et al., 2007; Semlitsch et al., 2015; Werner et al., 2007), although some highly specialized amphibian species have been associated with smaller wetlands (i.e. *L. sylvaticus* < 0.25 ha, Babbitt, 2005; Werner et al., 2007). The created wetlands we studied at MNF are smaller (≤ 0.1 ha) than this size range. Although wetland surface area was not an indicator of amphibian occupancy at our study site, it was a predictor of *A. maculatum* and *N. viridescens* abundance with a marked increase in predicted abundance at wetland surface areas >50 m² (0.005 ha). Denton and Richter (2013) also documented higher abundances (catch per unit effort) for these same species with an increase in wetland size, ranging 16–613 m² (0.0016–0.0613 ha). Larger wetlands can decrease inter- and intra-dependent competition and overall stress; for example, Millikin et al. (2019) found lower water-borne corticosterone levels, and therefore lower stress levels, in larval *A. maculatum* in wetlands with a larger diameter from our same study site. Creating larger wetlands (surface areas >50 m²) should be a consideration for salamander species conservation at the MNF site.

Landscape placement of wetlands is an important consideration when creating wetlands. Previous research has suggested a positive relationship between habitat connectivity and amphibian occupancy and colonization of wetlands (Hamer et al., 2016; Rannap et al., 2009). Laan and Verboom (1990) found that distance to nearest woodland was an important predictor of amphibian utilization of created wetlands. Likewise, Rannap et al. (2009) documented an affinity for created and restored forested wetlands by the Crested Newt (*Triturus cristatus*). This may be particularly important for salamanders as multiple stages of their life require upland forest (Laan and Verboom, 1990). We hypothesized distance to nearest forest would be a predictor of amphibian occupancy and abundance as the MNF site prior to mining and logging activity was heavily forested. Interestingly, four of the five species we studied did not have any association with distance to forest; only *N. viridescens* abundance was greater with a shorter distance to forest. As stated previously, the MNF site was prepared by leaving downed trees, logs, and rocks as upland and within-wetland habitat. In addition, the MNF site has high wetland habitat connectivity (>1400 wetlands). It is likely that these attributes of the restoration mitigated the immediate need for forest cover for many of these amphibian species by providing abundant cover objects and moist microclimates adjacent to the created wetlands. Roe and Grayson (2008) tracked *N. viridescens* terrestrial overnight movements and found the underside of leaves on the forest floor to be the primary microhabitat use for adults and efts at 40.2% and downed branches at 29.5% of all microhabitat use. While MNF site preparation methods account for the addition of downed branches, a leaf-litter layer is difficult to replicate without mature trees. Downed branches in the modified MNF landscape may facilitate occupancy of *N. viridescens* in the created wetlands while lack of a leaf litter layer may deter some *N. viridescens* movement leading to lower abundances. Importance of forested areas in close proximity to or surrounding wetlands is regional and based on historical landscapes. For example, Terrell et al. (2014) and Stiles et al. (2017), documented substantial amphibian use (14 species, 15,844 individuals over two years) of two wetlands embedded in a restored grassland on a surface mine, historically eastern deciduous forest with interspersed prairies. We suggest land managers consider the historic landscape of a restoration site when deciding on wetland placement.

Amphibians osmoregulate through their skin and are sensitive to changes in water chemistry within their habitats (Ultsch et al., 1999). Water quality parameters at our precipitation-fed study wetlands were within a normal range for both created and natural wetlands (Kutka and Bachmann, 1990; Drayer et al., 2020; Brodman et al., 2003; Whigham and Jordan, 2003; Byers et al., 2007), and within those normal

parameters, our wetland water quality varied by wetland age. These small variances in water quality parameters are likely not biologically relevant. Here we discuss, in the context of our results, three water quality parameters known to impact amphibian development: conductivity, sulfate, and pH, all associated with mining affected areas. Mean conductivity and sulfate values observed at MNF sites were several magnitudes lower (conductivity 6×, sulfate 100×) than those referenced in stream studies where reduced amphibian occupancy and abundance rates were reported (Hutton et al., 2020; Muncy et al., 2014; Price et al., 2016). Mean conductivity for all wetland age classes at the MNF site was comparable to peatland wetlands in New York (100.8–352.2 mg L⁻¹, Whigham and Jordan, 2003) and below the U.S. EPA's conductivity benchmark of 300 μS cm⁻¹ for aquatic life in central Appalachia (US EPA, 2011). Even though sulfate levels in the 8 year-old wetlands were more than double the mean of younger wetlands at MNF, sulfate mean values were lower than wetlands in Indiana (18.8–27.5 mg L⁻¹, Brodman et al., 2003), comparable to Kentucky (2.11–3.05 mg L⁻¹, Drayer et al., 2020), and far below the reported levels (206.37–853.61 mg L⁻¹) found in Appalachian streams affected by surface mining (Hutton et al., 2020, Muncy et al., 2014, Price et al., 2016). While mean pH values in our wetlands seem low (3.20–4.89) in comparison with wetlands in other areas (4.55–6.37, Ohio, Clark, 1986; 5.1–5.9, Wisconsin, Kutka and Bachmann, 1990; 5.09–5.33, Kentucky, Drayer et al., 2020), wetlands in conifer dominated forests can exhibit naturally occurring pH measurements as low as 3.0 (Whigham and Jordan, 2003, Byers et al., 2007) due to decomposition of acidic conifer needles deposited into the water column (Deano and Robinson, 1985; Hughes, 2018). For this reason, the higher percent canopy cover in the 8 year-old wetlands likely contributed to the lowest measurements we observed for pH. In addition, there is some evidence to suggest amphibians in conifer dominated habitats have a naturally high tolerance to low pH (Otto et al., 2007).

Our research supports a growing body of evidence for use of created wetlands in anthropogenically disturbed landscapes as mitigation for amphibian habitat loss (Birx-Raybuck et al., 2010; Drayer et al., 2020; Lannoo et al., 2009; Loughman, 2005; Stiles et al., 2017; Terrell et al., 2014). Our occupancy and abundance results are similar to created and natural wetlands in other regions where these species occur. Our higher end occupancy estimates for *A. maculatum* (0.82) and *L. clamitans* (1.00) are comparable to mean estimates in created wetlands in Kentucky, USA (*A. maculatum* = 0.81, *L. clamitans* = 0.96, Drayer et al., 2020) and natural wetlands in Ontario, Canada (*L. clamitans* = 0.73–0.83, Hecnar and M'Closkey, 1997); while our lower range occupancy estimates (*A. maculatum* = 0.56, *L. clamitans* = 0.26) are more similar to natural wetland sites in Maryland, USA (*A. maculatum* = 0.19–0.51, *L. clamitans* = 0.31–0.72, Zipkin et al., 2012) and regional occupancy estimates in Ontario, Canada (*L. clamitans* = 0.47–0.80, Hecnar and M'Closkey, 1997). Likewise, our abundance estimate for *A. maculatum* (12.7 individuals at wetland size = 69 m²) is similar to created wetlands in Kentucky (*A. maculatum* = 13.23 individuals, Drayer et al., 2020). Although our amphibian occupancy and abundance estimates are similar to other sites, amphibian occupancy and abundance of wetlands may fluctuate annually based on environmental conditions and competition pressures (Lannoo and Stiles, 2020). Our research, therefore, should be considered based on the limits of conclusions drawn on a single season.

5. Conclusions

The success of created wetlands at MNF as amphibian habitat demonstrates the potential for creating wetlands on reforested surface mine sites throughout the Appalachian Mountains, provided site preparation mirrors efforts at MNF (i.e. leaving downed trees, logs, and rocks within and adjacent to created wetlands and creating a high density matrix of wetlands). Using materials already present on the site is cost-effective and easily implemented alongside site preparation for tree plantings and, because amphibian occupancy is not tied to vegetation presence,

plantings can be phased. A few amphibian species at the MNF site could benefit from vegetation removal at densely vegetated wetlands in late autumn, especially vegetation species which produce litter high in C:N ratios, N:P ratios, and soluble phenolics (Cohen et al., 2012; Harrison et al., 2017; Rannap et al., 2020). Management for salamander species, *A. maculatum* and *N. viridescens*, at MNF is more nuanced as larger wetland sizes (50m²–121m²) benefit both species by increasing estimated abundances, and further, *N. viridescens* abundance is aided by in-wetland vegetation cover and closer proximity to forest. Similar to Petranka and Holbrook (2006), we suggest creating a matrix of wetlands with varying sizes, vegetative attributes, and proximity to forest to fully encompass all amphibian species present at resurfaced mine sites.

Funding

This work was supported by the University of Kentucky (UK) Graduate School, UK Department of Forestry and Natural Resources, UK Appalachian Center (Eller and Billings Research Award), UK College of Agriculture, Food, and Environment (Karri Casner Environmental Science Fellowship), the Chicago Herpetological Society, and the United States Department of Agriculture McIntire Stennis Cooperative Forestry Research Program (accession number 1001968). This research was approved under UK Institutional Animal Care and Use Committee protocol (2013-1073). Funding providers played no role in the formation of study design, data collection, data analysis and interpretation, writing of the report, and the decision to submit the article for publication.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

We thank the U.S. Forest Service Monongahela National Forest, especially Anna Branduzzi and Todd Kuntz, as well as Thomas Skinner, and Kylie Schmidt for help in the field. John Cox provided guidance and Allison Davis assisted with ArcGIS. Research collection permits were provided by the West Virginia Division of Natural Resources (2018.208).

References

- Andersen, C.P., Bussler, B.H., Chaney, W.R., Pope, P.E., Byrnes, W.R., 1989. Concurrent establishment of ground cover and hardwood trees on reclaimed mined land and unmined reference sites. *For. Ecol. Manag.* 28 (2), 81–99. [https://doi.org/10.1016/0378-1127\(89\)90062-5](https://doi.org/10.1016/0378-1127(89)90062-5).
- Babbitt, K.J., 2005. The relative importance of wetland size and hydroperiod for amphibians in southern New Hampshire, USA. *Wet. Ecol. Manag.* 13 (3), 269–279. <https://doi.org/10.1007/s11273-004-7521-x>.
- Batzler, D.P., Sharitz, R.R. (Eds.), 2014. *Ecology of Freshwater and Estuarine Wetlands*. University of California Press.
- Bernhardt, E.S., Palmer, M.A., 2011. The environmental costs of mountaintop mining valley fill operations for aquatic ecosystems of the Central Appalachians. *Ann. N. Y. Acad. Sci.* 1223 (1), 39–57. <https://doi.org/10.1111/j.1749-6632.2011.05986.x>.
- Biebighauser, T.R., 2011. *Wetland Restoration and Construction: A Technical Guide*. Upper Susquehanna Coalition, Owego.
- Birx-Raybuck, D.A., Price, S.J., Dorcas, M.E., 2010. Pond age and riparian zone proximity influence anuran occupancy of urban retention ponds. *Urban Ecosyst.* 13 (2), 181–190. <https://doi.org/10.1007/s11252-009-0116-9>.
- Bishop, S.C., 1941. The salamanders of New York. In: *New York State Museum Bulletin*, Number 324, Albany, New York.
- Brodman, R., Ogger, J., Bogard, T., Long, A.J., Pulver, R.A., Mancuso, K., Falk, D., 2003. Multivariate analyses of the influences of water chemistry and habitat parameters on the abundances of pond-breeding amphibians. *J. Freshw. Ecol.* 18 (3), 425–436. <https://doi.org/10.1080/02705060.2003.9663978>.
- Brodman, R., Parrish, M., Kraus, H., Cortwright, S., 2006. Amphibian biodiversity recovery in a large-scale ecosystem restoration. *Herpetol. Conserv. Biol.* 1 (2), 101–108.
- Brown, D.J., Street, G.M., Nairn, R.W., Forstner, M.R., 2012. A place to call home: amphibian use of created and restored wetlands. *Int. J. Ecol.* 2012 (2012), 11. <https://doi.org/10.1155/2012/989872>.

- Burger, J., Graves, D., Angel, P., Davis, V., Zipper, C., 2005. The forestry reclamation approach. In: US Office of Surface Mining Forest Reclamation Advisory No. 2. <https://doi.org/10.1007/s00267-011-9670-z>.
- Burnham, K.P., Anderson, D.R., 2002. Information and likelihood theory: a basis for model selection and inference. In: *Model Selection and Multimodel Inference: A Practical Information-theoretic Approach*, pp. 49–97. https://doi.org/10.1007/978-0-387-22456-5_2.
- Byers, E.A., Vanderhorst, J.P., Streets, B.P., 2007. *Classification and Conservation Assessment of High Elevation Wetland Communities in the Allegheny Mountains of West Virginia*. WV Natural Heritage Program, WVDNR, Wildlife Resources Section, Elkins, WV.
- Byers, E.A., Vanderhorst, J.P., Streets, B.P., 2010. *Classification and Conservation Assessment of Upland Red Spruce Communities in West Virginia*. WV Natural Heritage Program, WVDNR, Wildlife Resources Section, Elkins, WV.
- Calhoun, A.J., Arrigoni, J., Brooks, R.P., Hunter, M.L., Richter, S.C., 2014. Creating successful vernal pools: a literature review and advice for practitioners. *Wetlands* 34 (5), 1027–1038. <https://doi.org/10.1007/s13157-014-0556-8>.
- Clark, K.L., 1986. Distributions of anuran populations in central Ontario relative to habitat acidity. *Water Air Soil Pollut.* 30 (3–4), 727–734. <https://doi.org/10.1890/11-0078.1>.
- Cohen, J.S., Maerz, J.C., Blossey, B., 2012. Traits, not origin, explain impacts of plants on larval amphibians. *Ecol. Appl.* 22 (1), 218–228. <https://doi.org/10.1890/11-0078.1>.
- Dahl, T.E., 2005. *Status and Trends of Wetlands in the Conterminous United States 1998 to 2004*. US Department of the Interior, US Fish and Wildlife Service, Fisheries and Habitat Conservation, Washington D.C.
- Dahl, T.E., 2011. *Status and trends of wetlands in the conterminous United States 2004 to 2009*. US Department of the Interior, US Fish and Wildlife Service, Fisheries and Habitat Conservation, Washington D.C.
- Deano, P.M., Robinson, J.W., 1985. The effect of decaying leaves on the pH and buffer capacity of waters. *J. Environ. Sci. Health A* 20 (8), 903–911. <https://doi.org/10.1080/10934528509375266>.
- Denton, R.D., Richter, S.C., 2013. Amphibian communities in natural and constructed ridge top wetlands with implications for wetland construction. *J. Wildl. Manag.* 77 (5), 886–896. <https://doi.org/10.1002/jwmg.543>.
- Dormann, C.F., Elith, J., Bacher, S., Buchmann, C., Carl, G., Carré, G., Marquéz, J.R.G., Gruber, B., Lafourcade, B., Leitao, P.J., 2013. Collinearity: a review of methods to deal with it and a simulation study evaluating their performance. *Ecography* 36, 27–46. <https://doi.org/10.1111/j.1600-0587.2012.07348.x>.
- Drayer, A.N., Richter, S.C., 2016. Physical wetland characteristics influence amphibian community composition differently in constructed wetlands and natural wetlands. *Ecol. Eng.* 93 (2016), 166–174. <https://doi.org/10.1016/j.ecoleng.2016.05.028>.
- Drayer, A.N., Guzy, J.C., Caro, R., Price, S.J., 2020. Created wetlands managed for hydroperiod provide habitat for amphibians in western Kentucky. *USA Wetlands Ecol. Manag.* 28, 543–558. <https://doi.org/10.1007/s11273-020-09730-3>.
- ESRI, 2013. ArcMAP Desktop, Release 10.5.1. Environmental Systems Research Institute, Redlands, CA.
- Fiske, I., Chandler, R., 2011. Unmarked: an R package for fitting hierarchical models of wildlife occurrence and abundance. *J. Stat. Softw.* 43 (10), 1–23. <https://doi.org/10.18637/jss.v043.i10>.
- Franklin, J.A., Zipper, C.E., Burger, J.A., Skousen, J.G., Jacobs, D.F., 2012. Influence of herbaceous ground cover on forest restoration of eastern US coal surface mines. *New For.* 43 (5–6), 905–924. <https://doi.org/10.1007/s11056-012-9342-8>.
- French, M., Barton, C.D., 2020. *Mined Land Restoration in the Monongahela National Forest 2010–2020*. Project Report. <https://www.greenforestwork.org/project-reports>.
- Gallant, A.L., Klaver, R.W., Casper, G.S., Lannoo, M.J., 2007. Global rates of habitat loss and implications for amphibian conservation. *Copeia* 2007 (4), 967–979. [https://doi.org/10.1643/0045-8511\(2007\)7\[967:GROHLA\]2.0.CO;2](https://doi.org/10.1643/0045-8511(2007)7[967:GROHLA]2.0.CO;2).
- Gibbons, J.W., Winne, C.T., Scott, D.E., Willson, J.D., Glaudas, X., Andrews, K.M., Todd, B.D., Fedewa, L.A., Wilkinson, L., Tsaligaris, S.J., Harper, S.J., Greene, J.L., Tuberville, T.D., Metts, B.S., Dorcas, M.E., Nestor, J.P., Young, C.A., Akre, A., Reed, R.N., Buhlmann, K.A., Norman, J., Croshaw, D.A., Hagen, C., Rothermel, B.B., 2006. Remarkable amphibian biomass and abundance in an isolated wetland: implications for wetland conservation. *Conserv. Biol.* 20 (5), 1457–1465. <https://doi.org/10.1111/j.1523-1739.2006.00443.x>.
- Grant, E.H.C., Miller, D.A., Schmidt, B.R., Adams, M.J., Amburgey, S.M., Chambert, T., Cruickshank, S.S., Fisher, R.N., Green, D.M., Hossack, B.R., Johnson, P.T., Joseph, M. B., Rittenhouse, T.A.G., Ryan, M.E., Waddle, H., Walls, S.C., Bailey, L.L., Fellers, G. M., Gorman, T.A., Ray, A.M., Pilloid, D.S., Price, S.J., Saenz, D., Sadinski, W., Muths, E., 2016. Quantitative evidence for the effects of multiple drivers on continental-scale amphibian declines. *Sci. Rep.* 6 (1), 1–9. <https://doi.org/10.1016/j.biocon.2019.03.003>.
- Green, D.M., Lannoo, M.J., Lesbarrères, D., Muths, E., 2020. Amphibian population declines: 30 years of progress in confronting a complex problem. *Herpetologica* 76 (2), 97–100. <https://doi.org/10.1655/0018-0831-76.2.97>.
- Greenberg, A.E., Clesceri, L.S., Eaton, A.D., 1992. *Standard Methods for the Examination of Water and Wastewater*. American Public Health Association.
- Hamer, A.J., Heard, G.W., Urlus, J., Ricciardello, J., Schmidt, B., Quin, D., Steele, W.K., 2016. Manipulating wetland hydroperiod to improve occupancy rates by an endangered amphibian: modelling management scenarios. *J. Appl. Ecol.* 53 (6), 1842–1851. <https://doi.org/10.1111/1365-2664.12729>.
- Hanski, I., Gilpin, M., 1991. Metapopulation dynamics: brief history and conceptual domain. *Biol. J. Linn. Soc.* 42 (1–2), 3–16. <https://doi.org/10.1111/j.1095-8312.1991.tb00548.x>.
- Harrison, M.M., Tyler, A.C., Hellquist, C.E., Pagano, T., 2017. Phenolic content of invasive and non-invasive emergent wetland plants. *Aquat. Bot.* 136, 146–154. <https://doi.org/10.1016/j.aquabot.2016.09.013>.
- Hecnar, S.J., M'Closkey, R.T., 1997. Spatial scale and determination of species status of the green frog. *Conserv. Biol.* 11 (3), 670–682. <https://doi.org/10.1046/j.1523-1739.1997.96096.x>.
- Hughes, J., 2018. *Freshwater Ecology and Conservation: Approaches and Techniques*. Oxford University Press. <https://doi.org/10.1093/oso/9780198766384.001.0001>.
- Hutton, J.M., Price, S.J., Bonner, S.J., Richter, S.C., Barton, C.D., 2020. Occupancy and abundance of stream salamanders along a specific conductance gradient. *Freshwater Sci.* 39, 433–446. <https://doi.org/10.1086/709688>.
- Kéry, M., Royle, J.A., 2016. *Applied Hierarchical Modeling in Ecology: Analysis of Distribution, Abundance and Species Richness in R and BUGS*, Vol. 1. Academic Press, Amsterdam.
- Kross, C.S., Richter, S.C., 2016. Species interactions in constructed wetlands result in population sinks for wood frogs (*Lithobates sylvaticus*) while benefiting eastern newts (*Notophthalmus viridescens*). *Wetlands* 36 (2), 385–393. <https://doi.org/10.1007/s13157-016-0751-x>.
- Kudray, G.M., Schemm, T., 2008. *Wetlands of the Bitterroot Valley: Change and Ecological Functions*. Montana Natural Heritage Program. <https://doi.org/10.5962/bhl.title.56942>.
- Kutka, F.J., Bachmann, M.D., 1990. Acid sensitivity and water chemistry correlates of amphibian breeding ponds in northern Wisconsin, USA. *Hydrobiologia* 208 (3), 153–160. <https://doi.org/10.1007/BF00007781>.
- Laan, R., Verboom, B., 1990. Effects of pool size and isolation on amphibian communities. *Biol. Conserv.* 54 (3), 251–262. [https://doi.org/10.1016/0006-3207\(90\)90055-T](https://doi.org/10.1016/0006-3207(90)90055-T).
- Lannoo, M., 2005. *Amphibian Declines: The Conservation Status of United States Species*. University of California Press. <https://doi.org/10.1525/california/9780520235922.001.0001>.
- Lannoo, M.J., Stiles, R.M., 2020. Uncovering shifting amphibian ecological relationships in a world of environmental change. *Herpetologica* 76 (2), 144–152. <https://doi.org/10.1655/0018-0831-76.2.144>.
- Lannoo, M.J., Kinney, V.C., Heemeyer, J.L., Engbrecht, N.J., Gallant, A.L., Klaver, R.W., 2009. Mine spoil prairies expand critical habitat for endangered and threatened amphibian and reptile species. *Diversity* 1 (2), 118–132. <https://doi.org/10.3390/d1020118>.
- Lehtinen, R.M., Galatowitsch, S.M., 2001. Colonization of restored wetlands by amphibians in Minnesota. *Am. Midl. Nat.* 145 (2), 388–396. [https://doi.org/10.1674/0003-0031\(2001\)145\[0388:CORWBA\]2.0.CO;2](https://doi.org/10.1674/0003-0031(2001)145[0388:CORWBA]2.0.CO;2).
- Leroux, S.J., 2019. On the prevalence of uninformative parameters in statistical models applying model selection in applied ecology. *PLoS One* 14, e0206711. <https://doi.org/10.1371/journal.pone.0206711>.
- Loughman, Z.J., 2005. *Natural History and Conservation Biology of a Southern West Virginia Contour Surface Mine Reptile and Amphibian Community (Paper 709)*. [Masters Thesis, Marshall University]. Marshall Digital Scholar.
- MacKenzie, D.I., Nichols, J.D., Lachman, G.B., Droege, S., Royle, J.A., Langtimm, C.A., 2002. Estimating site occupancy rates when detection probabilities are less than one. *Ecology* 83, 2248–2255. [https://doi.org/10.1890/0012-9658\(2002\)083\[2248:ESORWD\]2.0.CO;2](https://doi.org/10.1890/0012-9658(2002)083[2248:ESORWD]2.0.CO;2).
- Marsh, D.M., Fegraus, E., Harrison, S., 1999. Effects of breeding pond isolation on the spatial and temporal dynamics of pond use by the tungara frog, *Physalaemus pustulosus*. *J. Anim. Ecol.* 68 (4), 804–814. <https://doi.org/10.1046/j.1365-2656.1999.00332.x>.
- Mazerolle, M.J., . AICcmodavg: Model Selection and Multimodel Inference Based on (Q) AIC (c). R Package Ver. 2.2–2. https://cran.r-project.org/package=AICcmodavg_consultéle, p. 2019.
- Millikin, A.R., Woodley, S.K., Davis, D.R., Anderson, J.T., 2019. Habitat characteristics in created vernal pools impact spotted salamander water-borne corticosterone levels. *Wetlands* 39 (4), 803–814. <https://doi.org/10.1007/s13157-019-01130-5>.
- Minkin, P., Ladd, R., 2003. *Success of Corps-required Wetland Mitigation in New England*. US Army Corps of Engineers, Boston MA, New England District, Concord, MA, 3 April.
- Morin, D.J., Yackulic, C.B., Diffendorfer, J.E., Lesmeister, D.B., Nielsen, C.K., Reid, J., Schaubert, E.M., 2020. Is your ad hoc model selection strategy affecting your multimodel inference? *Ecosphere* 11, e02997. <https://doi.org/10.1002/ecs2.2997>.
- Muncy, B.L., Price, S.J., Bonner, S.J., Barton, C.D., 2014. Mountaintop removal mining reduces stream salamander occupancy and richness in southeastern Kentucky (USA). *Biol. Conserv.* 180 (2014), 115–121. <https://doi.org/10.1016/j.biocon.2014.09.040>.
- Orizaola, G.E., Brana, F.L., 2003. Oviposition behaviour and vulnerability of eggs to predation in four newt species (genus *Triturus*). *Herpetol. J.* 13 (3), 121–124.
- Otto, C.R., Forester, D.C., Snodgrass, J.W., 2007. Influences of wetland and landscape characteristics on the distribution of carpenter frogs. *Wetlands* 27 (2), 261–269. [https://doi.org/10.1672/0277-5212\(2007\)27\[1261:OWALC\]2.0.CO;2](https://doi.org/10.1672/0277-5212(2007)27[1261:OWALC]2.0.CO;2).
- Palis, J.G., 2005. *Scaphiopus holbrookii* (Harlan, 1935) eastern spadefoot. In: Lannoo, M. (Ed.), *Amphibian Declines: The Conservation Status of United States Species*. University of California Press, pp. 511–512. <https://doi.org/10.1525/california/9780520235922.001.0001>.
- Petranka, J.W., Holbrook, C.T., 2006. Wetland restoration for amphibians: should local sites be designed to support metapopulations or patchy populations? *Restor. Ecol.* 14 (3), 404–411. <https://doi.org/10.1111/j.1526-100x.2006.00148.x>.
- Plass, W.T., 1982. *The impact of surface mining on the commercial forests of the United States*. Post-min. Product. Trees 31 (1982), 1–7.
- Porej, D., Hetherington, T.E., 2005. Designing wetlands for amphibians: the importance of predatory fish and shallow littoral zones in structuring of amphibian communities. *Wetl. Ecol. Manag.* 13 (4), 445–455. <https://doi.org/10.1007/s11273-004-0522-y>.

- Price, S.J., Muncy, B.L., Bonner, S.J., Drayer, A.N., Barton, C.D., 2016. Effects of mountaintop removal mining and valley filling on the occupancy and abundance of stream salamanders. *J. Appl. Ecol.* 53 (2), 459–468. <https://doi.org/10.1111/1365-2664.12585>.
- Price, S.J., Freytag, S.B., Bonner, S.J., Drayer, A.N., Muncy, B.L., Hutton, J.M., Barton, C. D., 2018. Mountaintop removal mining alters stream salamander population dynamics. *Divers. Distrib.* 24 (9), 1242–1251. <https://doi.org/10.1111/ddi.12760>.
- R Core Team, 2020. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. <https://www.R-project.org/>.
- Rannap, R., Lohmus, A., Briggs, L., 2009. Restoring ponds for amphibians: a success story. *Hydrobiologia* 634, 87–95. <https://doi.org/10.1007/s10750-009-9884-8>.
- Rannap, R., Kaart, M.M., Kaart, T., Kill, K., Uuemaa, E., Mander, Ü., Kasak, K., 2020. Constructed wetlands as potential breeding sites for amphibians in agricultural landscapes: a case study. *Ecol. Eng.* 158, 106077. <https://doi.org/10.1016/j.ecoeng.2020.106077>.
- Resetarits Jr., W.J., Wilbur, H.M., 1989. Choice of oviposition site by *Hyla chrysocelis*: role of predators and competitors. *Ecology* 70 (1), 220–228. <https://doi.org/10.2307/1938428>.
- Roe, A.W., Grayson, K.L., 2008. Terrestrial movements and habitat use of juvenile and emigrating adult eastern red-spotted newts, *Notophthalmus viridescens*. *J. Herpetol.* 42 (1), 22–30. <https://doi.org/10.1670/07-040.1>.
- Royle, J.A., 2004. N-mixture models for estimating population size from spatially replicated counts. *Biometrics* 60 (1), 108–115. <https://doi.org/10.1111/j.0006-341X.2004.00142.x>.
- Savage, W.K., Zamudio, K.R., 2005. *Ambystoma maculatum* (Shaw, 1802) spotted Salamander. In: Lannoo, M. (Ed.), *Amphibian Declines: The Conservation Status of United States Species*. University of California Press, pp. 621–627. <https://doi.org/10.1525/california/9780520235922.001.0001>.
- Semlitsch, R.D., 2008. Moving wetland mitigation towards conservation banking. *Natl. Wetlands Newslett.* 30 (5), 16.
- Semlitsch, R.D., Bodie, J.R., 1998. Are small, isolated wetlands expendable? *Conserv. Biol.* 12 (5), 1129–1133. <https://doi.org/10.1046/j.1523-1739.1998.98166.x>.
- Semlitsch, R.D., Peterman, W.E., Anderson, T.L., Drake, D.L., Ousterhout, B.H., 2015. Intermediate pond sizes contain the highest density, richness, and diversity of pond-breeding amphibians. *PLoS One* 10 (4), e0123055. <https://doi.org/10.1371/journal.pone.0123055>.
- Semlitsch, R.D., Walls, S.C., Barichivich, W.J., O'Donnell, K.M., 2017. Extinction debt as a driver of amphibian declines: an example with imperiled flatwoods salamanders. *J. Herpetol.* 51 (1), 12–18. <https://doi.org/10.1670/16-090>.
- Sena, K., Barton, C., Hall, S., Angel, P., Agouridis, C., Warner, R., 2015. Influence of spoil type on afforestation success and natural vegetative recolonization on a surface coal mine in Appalachia, United States. *Restor. Ecol.* 23 (2), 131–138. <https://doi.org/10.1111/rec.12164>.
- Shulse, C.D., Semlitsch, R.D., Trauth, K.M., Williams, A.D., 2010. Influences of design and landscape placement parameters on amphibian abundance in constructed wetlands. *Wetlands* 30 (5), 915–928. <https://doi.org/10.1007/s13157-010-0069-z>.
- Shulse, C.D., Semlitsch, R.D., Trauth, K.M., Gardner, J.E., 2012. Testing wetland features to increase amphibian reproductive success and species richness for mitigation and restoration. *Ecol. Appl.* 22 (5), 1675–1688. <https://doi.org/10.1890/11-0212.1>.
- Sjögren, P.E.R., 1991. Extinction and isolation gradients in metapopulations: the case of the pool frog (*Rana lessonae*). *Biol. J. Linn. Soc.* 42 (1–2), 135–147. <https://doi.org/10.1111/j.1095-8312.1991.tb00556.x>.
- Skousen, J., Zipper, C., Burger, J., Angel, P., Barton, C., 2011. Selecting topsoil substitutes for forestry mine soils. In: *The American Society of Mining and Reclamation Proceedings. Sciences Leading to Success*. Lexington, KY, pp. 591–609.
- Stiles, R.M., Swan, J.W., Klemish, J.L., Lannoo, M.J., 2017. Amphibian habitat creation on postindustrial landscapes: a case study in a reclaimed coal strip-mine area. *Can. J. Zool.* 95 (2), 67–73. <https://doi.org/10.1139/cjz-2015-0163>.
- Sullivan, J., 1993. *Picea rubens*. In: *Fire Effects Information System*. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. <https://www.fs.fed.us/database/feis/plants/tree/picrub/all.html> (Accessed 17 March 2021).
- Surface Mining Control and Reclamation Act of 1977, 1977. 30 U.S.C. 1201–1328.
- Terrell, V.C.K., Klemish, J.L., Engbrecht, N.J., May, J.A., Lannoo, P.J., Stiles, R.M., Lannoo, M.J., 2014. Amphibian and reptile colonization of reclaimed coal spoil grasslands. *J. North Am. Herpetol.* 59–68. <https://doi.org/10.17161/jnah.vi1.11895>.
- Tukey, J.W., 1949. Comparing individual means in the analysis of variance. *Biometrics* 5 (2), 99–114. <https://doi.org/10.2307/3001913>.
- U.S. Geological Survey, 2011. 20140331, NLCD 2011 Land Cover (2011 Edition). U.S. Geological Survey, Sioux Falls, SD.
- Ultsch, G.R., Bradford, D.F., Freda, J., 1999. *Physiology: coping with the environment*. In: McDiarmid, R.W., Altig, R. (Eds.), *Tadpoles: The Biology of Anuran Larvae*. University of Chicago Press, Chicago, pp. 189–214.
- US EPA (U.S. Environmental Protection Agency), 2011. *A Field-based Aquatic Life Benchmark for Conductivity in Central Appalachian Streams*. EPA/600/R-10/023F. National Center for Environmental Assessment, Office of Research and Development, U.S. Environmental Protection Agency, Washington, D.C.
- Vasconcelos, D., Calhoun, A.J., 2006. Monitoring created seasonal pools for functional success: a six-year case study of amphibian responses, Sears Island, Maine, USA. *Wetlands* 26 (4), 992–1003. [https://doi.org/10.1672/0277-5212\(2006\)26\[992:mcspfj\]2.0.co;2](https://doi.org/10.1672/0277-5212(2006)26[992:mcspfj]2.0.co;2).
- Wake, D.B., 1991. Declining amphibian populations. *Science* 253 (5022), 860–861. <https://doi.org/10.1126/science.253.5022.860>.
- Walter, J.A., Neblett, J.C., Atkins, J.W., Epstein, H.E., 2017. Regional-and watershed-scale analysis of red spruce habitat in the southeastern United States: implications for future restoration efforts. *Plant Ecol.* 218 (3), 305–316. <https://doi.org/10.1007/s11258-016-0687-5>.
- Wassens, S., Hall, A., Osborne, W., Watts, R.J., 2010. Habitat characteristics predict occupancy patterns of the endangered amphibian *Litoria raniformis* in flow-regulated flood plain wetlands. *Austr. Ecol.* 35 (8), 944–955. <https://doi.org/10.1111/j.1442-9993.2010.02106.x>.
- Werner, E.E., Skelly, D.K., Relyea, R.A., Yurewicz, K.L., 2007. Amphibian species richness across environmental gradients. *Oikos* 116 (10), 1697–1712. <https://doi.org/10.1111/j.0030-1299.2007.15935.x>.
- Whigham, D.F., Jordan, T.E., 2003. Isolated wetlands and water quality. *Wetlands* 23 (3), 541–549. [https://doi.org/10.1672/0277-5212\(2003\)023\[0541:iwawq\]2.0.co;2](https://doi.org/10.1672/0277-5212(2003)023[0541:iwawq]2.0.co;2).
- Zedler, J.B., Kercher, S., 2005. Wetland resources: status, trends, ecosystem services, and restorability. *Annu. Rev. Environ. Resour.* 30, 39–74. <https://doi.org/10.1146/annurev.energy.30.050504.144248>.
- Zipkin, E.F., Grant, E.H.C., Fagan, W.F., 2012. Evaluating the predictive abilities of community occupancy models using AUC while accounting for imperfect detection. *Ecol. Appl.* 22 (7), 1962–1972. <https://doi.org/10.1890/11-1936.1>.
- Zipper, C.E., Burger, J.A., Skousen, J.G., Angel, P.N., Barton, C.D., Davis, V., Franklin, J. A., 2011. Restoring forests and associated ecosystem services on Appalachian coal surface mines. *Environ. Manag.* 47 (5), 751–765. <https://doi.org/10.1007/s00267-011-9670-z>.
- Zuur, A., Ieno, E.N., Walker, N., Saveliev, A.A., Smith, G.M., 2009. *Mixed effects models and extensions in ecology with R*. Springer Sci. Bus. Med. <https://doi.org/10.1007/978-0-387-87458-6>.