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Evaluating the Use of Reclaimed Forests by Threatened, Endangered and Species of Concern on Appalachian Coal Mines

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Abstract

With the implementation of the Appalachian Regional Reforestation Initiative (ARRI) in the mid-2000s and adoption of the Forestry Reclamation Approach (FRA), tens of thousands of acres of forest now exist on reclaimed surface mines throughout the central Appalachian region. In addition, wetlands have been constructed on many reclaimed surface mines. These forests and wetlands may provide habitat for many forest-dependent wildlife species, including those considered threatened, endangered, and species of concern. The main objective of the project was to provide information on wildlife response to reforestation of surface mines, using the FRA, in West Virginia and Kentucky. Wildlife populations were surveyed across four land cover treatments: young (1-5 yr) FRA sites, old (8-23 yr) FRA sites, traditionally reclaimed sites, and non-mined forests in West Virginia and Kentucky in 2022 and 2023. Major findings suggest FRA implementation supported native bird species, particularly those species associated with young forests and included avian species of concern including Grasshopper Sparrow, Blue-winged Warbler and Prairie Warbler. Several bat species, such as the Tricolored bat that is currently proposed to be listed as endangered under the Endangered Species Act, foraged over created wetlands on reforested surface mines. The FRA coupled with wetland creation restored habitat for pond-breeding amphibians including species sensitive to forest loss. Finally, camera traps collected > 1500 terrestrial mammal observations from FRA sites. Major findings suggest that the implementation of ARRI and use of the FRA coupled with wetland creation on active and legacy mines can help to restore habitats that support diverse wildlife communities, including species that are threatened, endangered or of concern.

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Executive Summary

With the implementation of the Appalachian Regional Reforestation Initiative in the mid-2000s and adoption of the Forestry Reclamation Approach (FRA), tens of thousands of acres of forest now exist on reclaimed surface mines throughout the central Appalachian region. In addition, wetlands have been constructed on many reclaimed surface mines. These forests and wetlands may provide habitat for many wildlife species, including those considered threatened, endangered, and species of concern. However, few assessments of wildlife utilization of these reclaimed forests and wetlands have been conducted. The goal of this project is to provide information on wildlife response to reforestation of surface mines. A better understanding of wildlife responses will guide future reclamation practices. Our objectives included: 1) to compare community composition, species occupancy, and species abundance of wildlife across a chronosequence of FRA sites with embedded wetlands including mined, young forest (~1-5 yr-old), mined, maturing forest (~8-23 yr-old) and traditionally reclaimed mine land (> 40 yr-old), and 2) to compare community composition, species occupancy, and species abundance of wildlife in FRA sites with embedded wetlands to unmined, mature forests that also included wetlands. We hypothesized that reclaimed forests and embedded wetlands provide suitable habitat for regional biodiversity including species of concern such as amphibians, birds, and bats on surface coal mines in Appalachia. We focused our sampling efforts on the following groups of wildlife: (1) migratory songbirds, (2) bats, (3) wetland-breeding amphibians, (4) terrestrial salamanders, and (5) terrestrial mammals. We used a variety of methods to survey and monitor wildlife. Specifically, we conducted point counts for birds, dipnet samples for amphibians, coverboard sampling for salamanders, and data on abiotic attributes of these systems. Furthermore, we manually processed > 20,000 minutes of audio recordings and 1.2 million images captured from camera traps. Although we provide major findings below, this report is organized into 6 sections describing the results for specific wildlife taxa. Each section includes detailed introduction, methods, results, discussion, and conclusion sections.

Major findings of the study include:

1. We found that FRA treatments support native bird species, particularly those species associated with young forests. In particular, the FRA avian community assemblage included avian indicator species associated with young forests. While forests established using the FRA were still largely unoccupied by mature forest birds at 11-yrs post-reclamation, occupancy of avian species that are indicators of mature forests increased with FRA forest age. This suggests that continued succession of FRA forests may improve habitat availability for diverse breeding bird communities typical of Central Appalachian forests. Our research indicated that implementing the FRA on active and legacy mines may help to restore native forest cover that can meet the habitat needs of diverse forest bird communities. See Section 1 for more details on bird responses to FRA.
2. We found that bats are utilizing restored wetlands on reforested surface mines for foraging. Although most bats detected are considered common species, the Tri-colored Bat, which is currently petitioned for inclusion on the US Endangered Species list, was detected at sites in West Virginia and Kentucky. See Section 2 for more details on bat responses to FRA.
3. The FRA coupled with wetland creation restores habitat for pond-breeding amphibians within a relatively short timeframe (<10 yrs). The strong amphibian community response to FRA was likely due to the high density of created, seasonal wetlands resulting in more breeding opportunities; and site preparation techniques, including soil decompaction, woody debris loading, and tree planting, that promote microhabitats and microclimate refugia for amphibians during the non-breeding season, and facilitate movement (i.e., dispersal, migration) that leads to successful colonization of newly created wetlands. See Section 3 and Section 6 for more details.
4. We found that terrestrial salamander abundances were low on the FRA sites and these sites likely do not yet provide the conditions preferred by terrestrial salamanders. We expect salamander abundances to increase as forests mature. See Section 4 for more details on terrestrial salamander responses to FRA.
5. We detected 9 and 13 terrestrial mammals in West Virginia and Kentucky, respectively. White-tailed Deer were the most commonly detected species in both regions. We did not detect any species currently considered

for federal listing or priorities at the State-level. Notable species detected in West Virginia sites included Bobcat, Fisher and American Black Bear. In Kentucky, 3 Bobcats and 3 American Black Bears were detected; both species were detected in younger FRA or older FRA treatments. See Section 5 for more details.

Experimental

Study Sites

We surveyed wildlife at Monongahela NF (WV) across a chronosequence of FRA sites paired with traditional reclaimed (grassland) sites and reference (non-mined forest) sites (Figure 1.1). In Kentucky, we also surveyed wildlife across a chronosequence of FRA sites, with recently planted FRA sites on Pine Mountain (Ataya), older FRA sites on Laurel Fork Surface mine, and reference sites on the Daniel Boone National Forest (Figure 6.1). Each sampling site was centered on a created or natural wetland surrounded by reclaimed forest, traditionally reclaimed or reference forest (Figure 1.2). Wetlands are often foci of wildlife activity and allowed us to detect wildlife that use FRA forests and other study treatments. They provide foraging areas for bats and birds and breeding habitats for amphibians. Wetlands are also important watering holes, and habitat that harbors both plants and prey species important to a variety of mammals. More information on study site specifics can be found in the preceding sections (see below).

Survey Methods

At each sampling point we will use of sampling methods to survey wildlife (Figure I.2).

1. We used standard bird point counts to detect breeding birds at each sampling site. Each site was sampled on three occasions during the study period, with a 7-day interval between consecutive surveys. We performed point counts in the absence of rain and strong winds, starting no earlier than 15-min before sunrise, and ending no later than 10:00 am. Observers recorded all birds seen or heard. Section 1 provides additional details. See Section 1 for more details on bird survey methods.
2. We deployed automated recording units (ARU) at each sampling site. ARUs are widely used to acoustically detect wildlife. Numerous studies have used ARUs to detect songs of migratory songbirds, bat echolocation calls and vocalizations of breeding frogs and toads. ARUs were set to continuously record sounds every 5 minutes from late January through July to capture periods of peak bird, bat, and amphibian activity. After the recordings were retrieved, data were transcribed manually (i.e., by human ear). See Figure I.1. Section 2 and 6 provides additional details.
3. We used aquatic surveys to assess amphibian use of wetlands across our forest treatments. Salamanders represent an ecologically important component of forests and wetlands in central Appalachian ecosystems. They do not vocalize; thus, ARUs were not effective at detecting their presence or abundance. Specifically, we used quantitative larval amphibian surveys (i.e. dipnet sweeps) to assess amphibian species richness, species' occupancy, species' abundance at each sampling point. Three surveys were conducted at each wetland from mid- May through July. Before each aquatic survey, we recorded variables expected to influence detection including water temperature and day-of-year. Captured larvae were counted, measured, classified to species, and released. See Section 3.



Figure I.1. Automated Recording Units (ARU) showing size and deployment. ARUs were used to monitoring birds, bats and amphibians are study sites.

4. We used standard terrestrial survey method using coverboards to assess terrestrial amphibians near each sample point. Coverboards are effective at monitoring amphibians because they mimic conditions (high moisture, low ambient air temperatures) required by salamanders and frogs when they occupy the terrestrial environment. Boards were checked multiple times from April through July. Captured amphibians were counted, measured, weighed and released at their capture location. See Section 4.
5. We deployed motion sensitive camera traps to detect non-vocal birds and mammals. We mounted cameras on 1.5-m tall stakes positioned ~1 m from wetlands. See Section 5.
6. In addition to the wildlife sampling data collected, we deployed environmental data loggers at each sampling point to measure air temperature, relative humidity, water temperature at 30-min intervals. These data allowed us to quantify environmental differences among the study sites. Precipitation, barometric pressure, solar radiation, wind speed, gust speed, ground temperature, dew point, and light intensity measurements were collected using a weather station at Robinson Forest and MNF. These data were incorporated into analyses throughout Sections 1-6.

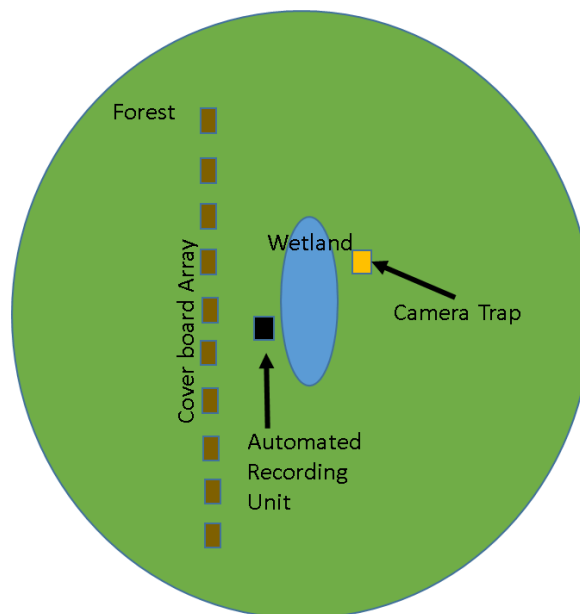


Figure I.2. Schematic of the survey methods used to monitor wildlife at our study sites.

Section 1. Effects of forest reclamation on avian community guilds and abundances occupying legacy surface mines

1.1 Introduction

Human disturbance can dramatically transform natural landscapes, often resulting in habitat fragmentation (Fischer and Lindenmayer 2007; Wilson et al. 2016), reductions in habitat quality and complexity (Caviedes and Ibarra 2017; Pike et al. 2010; Vásquez-Grandón, Donoso, and Gerding 2018), and changes in biodiversity (Barlow et al. 2016; Cardinale and Palmer 2002; Dornelas 2010). Surface coal mining is a large-scale human disturbance that has impacted about 600,000 ha of land across the Appalachian region (Zipper et al. 2011). Coal mining can disturb ecosystems by altering the landscape's natural topography, degrading forest habitat, and changing natural hydrology and soil properties (Wickham et al., 2013). Characterizing how mining and other resource extraction impacts distinct ecological communities is critical to inform local and regional management responses and conservation. In order to mitigate long-term shifts in community composition, it is essential for studies to monitor the downstream effects of disturbance and/or land restoration on local taxa. Surface mine reclamation aims to reduce the adverse environmental consequences of landscape change. Mine reclamation can generally be classified into two approaches: traditional reclamation into grassland, and the Forestry Reclamation Approach (FRA), developed in 2005. Under the Surface Mining Control and Reclamation Act (SMCRA) of 1977, coal mining companies are required to revegetate the landscape and restore the area to its approximate original contour to prevent soil erosion and landslides post-mining (Bulluck and Buehler 2006; Lambert et al. 2021; Burger et al. 2005). Traditional reclamation practices in the Appalachians typically involve soil compaction by heavy equipment and the planting of aggressive, non-native grasses, forbs, and shrubs to form grasslands (Angel et al. 2015). Despite the legislative intentions, traditional reclamation frequently results in poor hydrological function, limited root growth, loss of topographic complexity, forest loss, and loss of biodiversity (Angel et al. 2015; Wickham et al. 2013; Williamson and Barton 2020). Moreover, traditionally-reclaimed minelands often remain in a state of arrested succession, thus inhibiting successful reforestation of native trees (Bulluck and Buehler 2006; Burger et al. 2013; Zipper et al. 2011).

Contrary to traditional reclamation, the FRA aims to restore ecosystem function and native forest vegetation to minelands (Burger et al. 2005; Zipper et al. 2011). This approach was developed after extensive reclamation research, and involves a detailed five-step process for reclaiming active minelands to forest under SMCRA (Burger et al. 2005). Specifically, the FRA recommends methods for creating a suitable rooting medium, loosely grading topsoil, using native vegetation for erosion control ground-cover, and planting native tree species using proper planting techniques (Burger et al. 2005). Minelands reclaimed using the FRA have been observed to undergo accelerated natural succession compared to traditional methods that establish grasslands (Groninger et al. 2007). Consequently, the FRA may ultimately create conditions that maintain similar ecological communities to those of native Appalachian forests.

A modified form of the FRA was developed for “legacy mines,” or surface mines that were originally reclaimed to grassland or shrubland (Burger et al. 2013). Contrary to the FRA practices used on active mines, legacy mine FRA aims to reverse the damage from traditional reclamation through soil decompaction, non-native species removal, native tree and shrub planting, woody debris loading, and wetland creation (Burger et al. 2013). Despite their different applications, both forms of the FRA are expected to encourage the rapid succession of native forest and habitat provisioning for biotic communities. In particular, the FRA may benefit Appalachian forest-obligate bird species, many of which depend on contiguous multi-aged forest across their breeding range (Fiss et al. 2021; Schlossberg et al. 2018; Vitz and Rodewald 2006).

The FRA may differ from traditional reclamation practices in its capacity to support diverse forest bird communities, including many species experiencing dramatic population declines (Rosenberg et al. 2019). Theoretically, the FRA should benefit forest bird communities by initially providing a mix of herbaceous, shrub, and sapling cover known to be important to many forest birds during nesting and post-fledgling periods (Anders, Faaborg, and Thompson 1998; Fiss et al. 2021; Goguen 2019; King et al. 2006), and later providing habitat for mature forest species after several decades of succession (Groninger et al. 2007). Given the accelerated natural succession of FRA forests (Groninger et al. 2007), we should also expect FRA sites to begin providing habitat conditions for forest birds associated with closed-canopy (or older) forests during relatively early stages of forest reclamation. Nevertheless, no studies have verified these expectations by empirically examining the effects of the FRA on avian communities (Lituma et al. 2021; Zipper et al. 2011). This is a critical research gap, as FRA implementation could potentially contribute to avian conservation efforts across hundreds of thousands of hectares of land in the Appalachians, where the recovery of forest bird species is a regional priority (Franzreb and Phillips 1996; Franzreb and Rosenberg 1997).

One way to discern whether reclaimed landscapes are providing habitat for breeding forest birds is by comparing breeding season bird communities on reclaimed mines to those within native forests. At high elevations in the Central Appalachians, the native land cover has historically consisted of red spruce (*Picea rubens*)-dominated forests. Due to surface coal mining, unsustainable timber harvest, and wildfire damage, red spruce populations in the Central Appalachians have been significantly reduced and replaced by even-aged, northern hardwoods (Hopkins 1899; Schuler, Ford, and Collins 2002; Trani 2002). A recent study identified 11 bird species that are strong indicators of red spruce-northern hardwood (RS-NH) forests in the Central Appalachians (Clipp et al. 2022). The presence/absence of these indicator species on FRA minelands can therefore inform reclamation success for breeding bird communities.

We assessed the response of forest bird communities to minelands reclaimed using the FRA after approximately one decade of forest growth. To evaluate FRA outcomes for forest birds, our study had two main questions: First, how does the composition of breeding bird communities within FRA forests compare to those within older unmined and mined forests? In other words, do FRA forests and older forests provide habitat for a similar composition of avian nesting habitat guilds (e.g., mature forest, young forest, generalist)? Second, are RS-NH bird indicator species occupying forests created through the FRA? If so, how does their occupancy compare between FRA forests and older non-FRA forests? To answer these questions, we assessed the composition of breeding bird communities and abundances across four age classes of RS-NH forest: 1) younger FRA (2-5 yr), 2) older FRA (8-11 yr), 3) naturally regenerated forest on mineland (>40 yr), and 4) unmined mature forest (>100 yr). Due to the accelerated natural succession of FRA forests (Groninger et al. 2007), we hypothesized that the composition of breeding bird communities and abundances within FRA forests would show a similar pattern to communities found within forests naturally regenerated on mineland, but would greatly differ from those occupying unmined mature forests. We predicted that the FRA breeding bird community would include RS-NH indicator species, especially those that breed in young RS-NH forests. The results of this study may establish a framework for long-term studies of avian communities as FRA forests continue to age and potentially more closely resemble native RS-NH forests.

1.2. Methods

1.2.1. Study Area

Our study occurred in the Monongahela National Forest (MNF), located in the Allegheny Mountains of eastern West Virginia (USA) (Figure 1.1). The MNF contains over 371,906 ha across 10 counties in the West Virginia highlands, with elevations ranging from around 305 to 1,482 m above sea level (USDA Forest Service, n.d.). Common hardwood species include American beech (*Fagus grandifolia*), sugar maple (*Acer saccharum*), and black cherry (*Prunus serotina*) (Clipp et al. 2022). Additionally, red spruce is a dominant species within the MNF, found along high peaks and ridgelines (Rentch et al. 2007; Stephenson and Clovis 1983).

Within the MNF, we focused on legacy minelands at the Mower Tract and Sharp Knob, located in Randolph and Pocahontas Counties, West Virginia, respectively. The Mower Tract is a 16,187-ha, high-elevation area of the MNF located on Cheat Mountain (1478 m), and was purchased by the Mower Land and Lumber Company in the early 1980s (Green Forests Work 2020). In addition to intensive logging,

approximately 809 ha of the Mower Tract were surface mined for coal in the late 1970s and early 1980s and reclaimed to grasslands, further contributing to the loss of native RS-NH forests (Green Forests Work 2020). RS-NH forests were similarly degraded at Sharp Knob (1382 m), a mountain peak where coal mining persisted through the early 1970s, followed by traditional reclamation into grassland.

In 2008, the US Forest Service partnered with Green Forests Work and the Appalachian Regional Reforestation Initiative to begin implementing ecological restoration on legacy minelands in the MNF. From 2009 to 2020, Green Forests Work planted over 500,000 native trees across 400 ha of the MNF (Green Forests Work 2020). In addition to recommended FRA practices used on active mine sites (Burger et al. 2005), Green Forests Work used modified FRA techniques that pertain specifically to legacy mine sites (Burger et al. 2013), including non-native species removal and soil de-compaction using a deep ripping shank. Green Forests Work also created wetlands and loaded woody debris within the project areas to enhance structural complexity, provide wildlife habitat, slow surface runoff of water, and enhance depressional water storage on-site (Branduzzi, Barton, and Lovell 2020).

1.2.2. Treatment Types

To assess breeding community response to mine reclamation, we selected 32 sites within the MNF (29 in the Mower Tract and 3 at Sharp Knob) (Figure 1.2). Sites were divided equally among 4 treatment types: 1) younger FRA (YFRA), 2) older FRA (OFRA), 3) naturally regenerated forest on mineland (REGEN), and 4) unmined, mature forest (MAT) (Figure 1.2). YFRA sites were reclaimed between 2017 and 2020, or 2-5 years prior to data collection in 2022. These sites were planted with a seedling mix of red spruce and 12 hardwood species, including aspen (*Populus spp.*), black cherry (*Prunus serotina*), mountain ash (*Sorbus americana*), sugar maple (*Acer saccharum*), and serviceberry (*Amelanchier arborea*). In 2021, seedlings at YFRA sites had a mean height of 0.36 m (Rhodes 2022). At the time of data collection in 2022, planted trees and shrubs averaged <1 m in height. Generally, YFRA sites were characterized by extensive herbaceous groundcover, including species from the original mine reclamation mix and naturally colonized species that emerged after ripping (Branduzzi, A 2020). OFRA sites were reclaimed between 2011 and 2014, or 8-11 years before data collection. During site preparation, OFRA sites were initially planted with a seedling mix of red spruce and 3 hardwood species – aspen, black cherry, and serviceberry. In 2021, seedlings at OFRA sites had an average height of 2.01 m (Rhodes 2022). At the time of data collection in 2022, planted trees and shrubs had reached approximately 2-4 m in height. Based on the planting of herbaceous cover as well as early tree and shrub growth, both YFRA and OFRA sites were classified as young successional forest. Coal mining at REGEN sites occurred prior to 1977, or over 40 years prior to our study. Given that these sites were mined before SMCRA was passed, the land was not restored to its original contour and all REGEN sites were adjacent to an exposed rock face, or highwall. All of the REGEN benches (i.e., narrow, horizontal strips of land cut into the side of open-pit mines) were initially reclaimed as grasslands, and all of these sites were planted with non-native conifers such as Norway spruce (*Picea abies*). The long and narrow mine benches were surrounded, above and below, by native forest which could have provided a seed source for natural succession; however, the compacted nature of the soils, and the inability of Norway spruce to facilitate regeneration of other species, have slowed succession (Branduzzi, 2020). Regardless, the lengthy time since reclamation and narrowness of the mine benches at REGEN sites contributed to moderate recolonization and natural regeneration of red spruce and other native plants, resulting in a mosaic of native and exotic tree species with varying age classes amidst patches of grassland. We categorized REGEN sites as a mixture of herbaceous, small-diameter woody stems, and maturing forest. Finally, MAT sites were unmined patches of second-growth, RS-NH forest, harvested during the early 1900s. They were predominantly composed of red spruce, beech (*Fagus grandifolia*), yellow birch (*Betula alleghaniensis*), and red maple (*Acer rubrum*). MAT sites functioned as a reference treatment for the other three mineland treatments.

Each site was centered around either a created or natural wetland. This study was part of a large-scale assessment of wildlife response to the FRA that included birds, amphibians, bats, and other mammals. Given the close association between wetlands and many of the target species, wetlands were a necessary feature for site selection and were the centralized location for all wildlife surveying. Wetlands ranged in total area from 5 – 233 m². All selected wetlands were positioned at least 200 m apart to facilitate spatial independence between avian surveys.

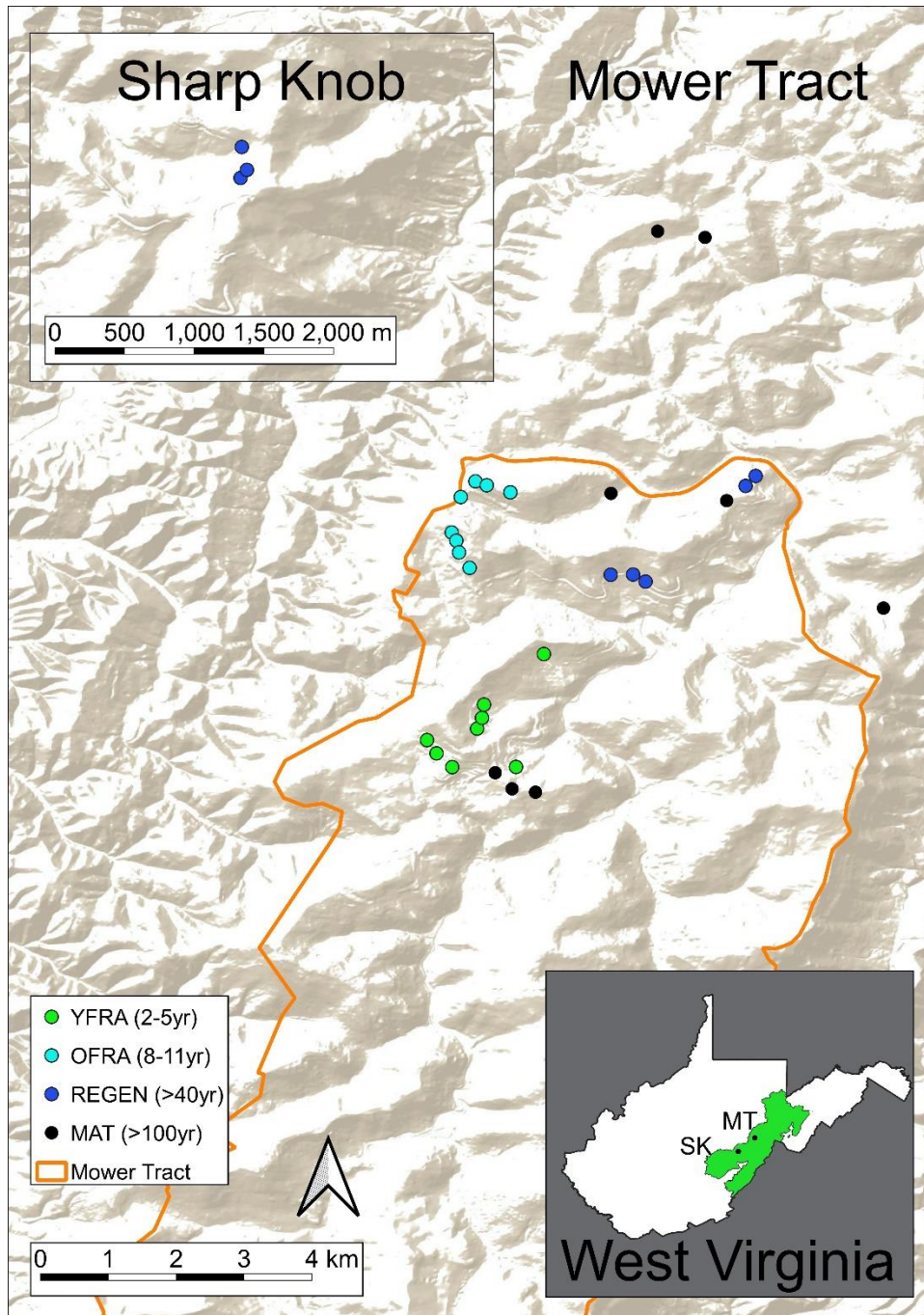


Figure 1.1. Map of 32 sites across the Monongahela National Forest (West Virginia, USA), each belonging to one of four treatment type categories. White sites are younger FRA sites that were reclaimed between 2-5 years ago (YFRA). Yellow sites represent older FRA sites that were reclaimed between 8-11 years ago (OFRA). Green sites represent naturally regenerated forests that were mined >40 years ago (REGEN). Brown sites indicate unmined, mature reference sites that were logged >100 years ago (MAT). 29 sites fall within the Mower Tract red boundary, while 3 sites are positioned within Sharp Knob.



Figure 1.2 Photographs of four treatment types: A) Younger (2-5 yr) FRA (YFRA), B) older (8-11 yr) FRA (OFRA), C) non-FRA, naturally regenerated minelands (> 40 yr) (REGEN), and D) unmined, mature forest (MAT).

1.2.3. Point Count Surveys

From June 1 through June 23, 2022, we surveyed breeding bird communities using 10-min, 100-m radius, stationary point count surveys. Each site was sampled on three occasions during the study period, with a 7-day interval between consecutive surveys. We performed point counts in the absence of rain and strong winds, starting no earlier than 15-min before sunrise, and ending no later than 10:00 am. Prior to each point count, we recorded day of year, time of survey, air temperature, wind speed, and cloud cover to include as survey covariates in our analyses. Cloud cover and wind speed were estimated using the Okta scale (Stull 2015) and Beaufort wind force scale (World Meteorological Organization 1970), respectively. Each survey was conducted simultaneously by two observers, both of whom were trained and tested to minimize bias and confirm accuracy. Observers recorded all species seen or heard during the survey, excluding all flyovers. If there were discrepancies between the two reports, we adjusted the data record to include all species confidently detected by either observer during the survey. If a species was only detected by one observer, we referred back to an audio recording taken during the 10-min survey on the mobile application software, Merlin (The Cornell

Lab 2014), to either confirm or deny the presence of the species. Audio recordings were only used to cross-reference species that had already been detected by human observers during the point count and did not otherwise supplement survey data.

1.2.4. Multi-Species Bayesian Community Occupancy Model

To assess the effect of reclamation approach and survey covariates on avian communities, we constructed a Bayesian hierarchical multi-species community occupancy model (MacKenzie et al. 2017; Zipkin, DeWan, and Andrew Royle 2009). This model incorporates site and survey covariates to provide species-specific estimates of occupancy and detection probability, and to estimate the mean community response (Dorazio and Royle 2005). Specifically, we estimated avian community response to categorical site-specific covariates of treatment type (i.e., YFRA, OFRA, REGEN or MAT). We also tested for meaningful associations between detection probability and five survey-level covariates: day of year (DOY), time of day (Time), estimated cloud cover (Cloud), estimated wind speed (Wind), and air temperature (Temp). We did not include wetland area as a site covariate given that wetland size and other site characteristics (i.e., distance to road and proportion of forest cover) did not have a meaningful relationship with avian occupancy (Davenport 2023).

Our community modeling approach combines single species estimates with the average parameter estimate for the entire community. This approach reduces bias and improves the precision of parameter estimates for species with few detections, or “data-poor” species, by borrowing information from species with higher detections, or “data-rich” species (Pacifiçi et al. 2014; Sauer and Link 2002). Therefore, we separated species into three distinct nesting habitat guilds: 1) mature forest, 2) young forest, and 3) generalist, based on previous classifications of their breeding biology (Byers, Vanderhorst, and Streets 2010; Canterbury et al. 2000). Both young forest and mature forest habitat guilds of birds have strongly positive within-group correlations, or similar population responses to environmental variation (Canterbury et al. 2000). Consequently, it is crucial that the community consists of species with either ecological, functional, or behavioral relatedness (Pacifiçi et al. 2014). By separating our pool of species into these functional groups (i.e., nesting guilds), we ensured that the models would only borrow data from species with shared nesting habitat requirements.

We formatted our data into a matrix (i,j,k) , where species i was detected at site j on sampling occasion k . Occupancy probability, Ψ_{ij} , represented the probability of species i occurring at site j , modeled as a function of covariate parameters on α . Detection probability, p_{ijk} , represented the probability of detecting species i at site j on sampling occasion k , modeled as a function of covariate parameters on β . We modeled multi-species occupancy for each guild with the following equation, applied to three models (i.e., mature forest, young forest, and generalist guilds):

$$\text{logit}(\Psi_{ij}) = u_i + \alpha_{1i}\text{YFRA}_j + \alpha_{2i}\text{OFRA}_j + \alpha_{3i}\text{REGEN}_j$$

We modeled detection probability for each guild with the following equation applied to three models (i.e., mature forest, young forest, and generalist guilds):

$$\text{logit}(p_{ijk}) = v_i + \beta_{1i}\text{DOY}_{jk} + \beta_{2i}\text{Time}_{jk} + \beta_{3i}\text{Cloud}_{jk} + \beta_{4i}\text{Wind}_{jk} + \beta_{5i}\text{Temp}_j$$

Each model contained a total of eight parameters: α_{1i} , α_{2i} , α_{3i} , β_{1i} , β_{2i} , β_{3i} , β_{4i} , and β_{5i} . Parameters α_1 , α_2 , and α_3 were effects of the categorical variable of treatment type, with “MAT” as the reference category. The remaining parameters were continuous covariates, standardized to have mean of zero.

We estimated species richness (SpR) at sampled sites by summing indicator variables for occupancy for each species at each site for each model iteration to generate a posterior predictive distribution for species richness in each treatment. More specifically, at each iteration, the model estimated a latent state variable (occupied or unoccupied) for species i at site j , which we summarized across all species (within the guild) at a given site for that iteration. We then calculated the mean species richness per treatment type for each guild.

We used a Bayesian framework with Markov chain Monte Carlo sampling (Link et al. 2002). We used a uniform distribution and uninformative priors with a mean of -3 and precision of 3 (i.e., $U(-3, 3)$) for α , β , and community-level parameters, and $U(0, 5)$ for all standard deviation (σ) parameters). The model was run with three parallel chains, which were checked for convergence using the Gelman-Rubin statistic; for all monitored

parameters in this study, this value was at or below 1.02, indicating convergence (Gelman and Rubin 1992). Each chain was run for 70,000 iterations with a 20,000-sample burn-in and a thinning rate of 3. We thus retained 50,000 samples from the posterior distribution and approximated posterior summary statistics for each parameter, including mean, SD, and 95% Bayesian credible intervals (CIs). CIs are defined by the 2.5 and 97.5% percentiles of the distribution. Parameter estimates with credible intervals that did not contain zero were interpreted as biologically meaningful covariates for occupancy and/or detection probability. We used the R package R2WinBUGS (Sturtz, Ligges, and Gelman 2005) to export data into WinBUGS and perform the Bayesian analysis. R and WinBUGS code were developed by Guzy et al. (2019).

1.2.5. Single-Species Abundance Models

We used binomial mixture models (Royle 2004) to examine the effect of treatment type (i.e., YFRA, OFRA, REGEN, and MAT) and survey covariates (i.e., DOY, Time, Temp, Wind, and Cloud) on species-specific avian abundance. Our field protocol aligned with the assumptions of binomial mixture models by consisting of 3 replicate counts at 32 spatially distinct sites (i) during temporally indexed surveys (j). Counts were modeled as independent outcomes of binomial sampling with index N_i and detection probability p_j . Local-level abundances (λ) were modeled with a Poisson distribution. We used a Poisson-regression formulation of local mean abundances to model heterogeneity in abundance among bird populations due to site-specific covariates (x_i), where $\log(\lambda_i) = \alpha_1 + \alpha_2 x_i$. We also modeled associations between survey covariates and p_i to identify sources of heterogeneity in detection probability, where $\text{logit}(p_{ij}) = \beta_1 + \beta_2 x_{ij}$. We used the equation below to model heterogeneity in species-specific abundance:

$$N_i | \lambda_i \sim \text{Poi}(\lambda_i) \\ \log(\lambda_i) = \beta_1 + \beta_2 \text{YFRA}_i + \beta_3 \text{OFRA}_i + \beta_4 \text{REGEN}_i$$

We used the equation below to model heterogeneity in per-individual detection probability:

$$\text{logit}(p_{ij}) = \alpha_1 + \alpha_2 \text{DOY}_j + \alpha_3 \text{Time}_j + \alpha_4 \text{Cloud}_j + \alpha_5 \text{Wind}_j + \alpha_6 \text{Temp}_j$$

1.3. Results

1.3.1 Point Count Surveys

We detected 49 species during avian point count surveys (Table 1). In total, we detected 35, 32, 30, and 36 species in the YFRA, OFRA, REGEN, and MAT treatments, respectively. Within the nesting habitat guilds, we detected 26 mature forest species, 16 young forest species, and 7 habitat generalist species (Table 1.1.). Across all treatments, we detected 15 avian indicator species of RS-NH forests (Clipp et al. 2022). The number of detected indicator species in the YFRA, OFRA, REGEN, and MAT treatments were 11, 13, 13, and 12, respectively (Table 1). Eleven indicator species belonged to the mature forest guild, three belonged to the young forest guild, and one belonged to the habitat generalist guild. Detection of young forest guild species was negatively associated with day of year ($\beta_1 = -0.51$; 95% CI = -0.99, -0.08). Remaining survey covariates did not have meaningful associations with avian nesting habitat guild detection.

1.3.2. Species Richness and Occupancy Estimates

1.3.2.1. Younger (2-5 yr) FRA Treatment

Mean species richness estimates for the mature forest, young forest, and habitat generalist guilds were 10.47 (95% CI = 6.63, 15.88), 11.86 (95% CI = 8.75, 15.13), and 4.40 (95% CI = 3.25, 5.63), respectively (Figure 1.3). Mean occupancy estimates for the mature forest, young forest, and habitat generalist guilds were 0.36 (95% CI = 0.15, 0.65), 0.79 (95% CI = 0.54, 0.94), and 0.76 (95% CI = 0.32, 0.97), respectively (Figure 1.4). Estimated mean occupancy of the mature forest guild was negatively associated with YFRA sites, whereas occupancy of the young forest and habitat generalist guilds were both positively associated with YFRA sites (Figure 1.5). Of the 15 avian indicator species of RS-NH forests detected in our study (Clipp et al. 2022), occupancy estimates for 10 species were negatively associated with YFRA sites, and four species were positively associated with YFRA sites (Table 1.2).

Table 1.1. Number of detections of all species (n = 60) from point count surveys, categorized into one of three habitat guilds: 1) mature forest, 2) disturbed (shrubland/forest edge), and 3) generalist. Red Spruce – Northern Hardwood indicator species (Clipp et al. 2022) are indicated via asterisk.

Habitat Guild	Species	Detections
Mature Forest	Blackburnian Warbler (<i>Dendroica fusca</i>)*	32
	Black-capped Chickadee (<i>Poecile atricapillus</i>)	51
	Black-throated Blue Warbler (<i>Dendroica caerulescens</i>)	20
	Black-throated Green Warbler (<i>Setophaga virens</i>)*	156
	Blue-headed Vireo (<i>Vireo solitarius</i>)*	39
	Brown Creeper (<i>Certhia americana</i>)	6
	Canada Warbler (<i>Wilsonia canadensis</i>)	26
	Eastern Wood Pewee (<i>Contopus virens</i>)	1
	Golden-crowned Kinglet (<i>Regulus satrapa</i>)*	43
	Hairy Woodpecker (<i>Leuconotopicus villosus</i>)	3
	Hermit Thrush (<i>Catharus guttatus</i>)*	67
	Hooded Warbler (<i>Wilsonia citrina</i>)	1
	Magnolia Warbler (<i>Setophaga magnolia</i>)*	84
	Northern Waterthrush (<i>Seiurus noveboracensis</i>)	11
	Ovenbird (<i>Seiurus aurocapilla</i>)	4
	Pileated Woodpecker (<i>Dryocopus pileatus</i>)	3
	Pine Siskin (<i>Carduelis pinus</i>)	1
	Red Crossbill (<i>Loxia curvirostra</i>)*	34
	Red-bellied Woodpecker (<i>Melanerpes carolinus</i>)	2
	Red-breasted Nuthatch (<i>Sitta canadensis</i>)*	38
	Red-eyed Vireo (<i>Vireo olivaceus</i>)	46
	Rose-breasted Grosbeak (<i>Pheucticus ludovicianus</i>)*	1
	Ruffed Grouse (<i>Bonasa umbellus</i>)	1
	Scarlet Tanager (<i>Piranga olivacea</i>)	3
	Sharp-shinned Hawk (<i>Accipiter striatus</i>)	1
	Swainson's Thrush (<i>Catharus ustulatus</i>)*	48
Veery (<i>Catharus fuscescens</i>)	16	
White-breasted Nuthatch (<i>Sitta carolinensis</i>)	1	
Winter Wren (<i>Troglodytes hiemalis</i>)*	16	
Yellow-rumped Warbler (<i>Setophaga coronata</i>)*	49	
Yellow-throated Vireo (<i>Vireo flavifrons</i>)	3	
Disturbed	Alder Flycatcher (<i>Empidonax alnorum</i>)	13
	Brown Thrasher (<i>Toxostoma rufum</i>)	3
	Chestnut-sided Warbler (<i>Setophaga pensylvanica</i>)*	25
	Chipping Sparrow (<i>Spizella passerina</i>)	45
	Common Yellowthroat (<i>Geothlypis trichas</i>)	44
	Dark-eyed Junco (<i>Junco hyemalis</i>)*	164
	Eastern Towhee (<i>Pipilo erythrophthalmus</i>)	84
	Field Sparrow (<i>Spizella pusilla</i>)	71
	Indigo Bunting (<i>Passerina cyanea</i>)	83
	Mourning Warbler (<i>Geothlypis philadelphia</i>)*	5

	Red-winged Blackbird (<i>Agelaius phoeniceus</i>)	8
	Ruby-throated Hummingbird (<i>Archilochus colubris</i>)	4
	Song Sparrow (<i>Melospiza melodia</i>)	45
	Swamp Sparrow (<i>Melospiza georgiana</i>)	18
	White-throated Sparrow (<i>Zonotrichia albicollis</i>)	2
	Wild Turkey (<i>Meleagris gallopavo</i>)	1
	Yellow Warbler (<i>Setophaga petechia</i>)	2
Generalist	American Crow (<i>Corvus brachyrhynchos</i>)	44
	American Goldfinch (<i>Spinus tristis</i>)	8
	American Robin (<i>Turdus migratorius</i>)	140
	Blue Jay (<i>Cyanocitta cristata</i>)	54
	Carolina Wren (<i>Thryothorus ludovicianus</i>)	2
	Cedar Waxwing (<i>Bombycilla cedrorum</i>)	18
	Chimney Swift (<i>Chaetura pelagica</i>)	12
	Common Raven (<i>Corvus corax</i>)	1
	House Wren (<i>Troglodytes aedoni</i>)	8
	Mourning Dove (<i>Zenaida macroura</i>)	1
	Northern Flicker (<i>Colaptes auratus</i>)*	8
	Turkey Vulture (<i>Cathartes aura</i>)	4

1.3.2.2. Older (8-11 yr) FRA Treatment

Mean species richness estimates for the mature forest, young forest, and habitat generalist guilds were 10.22 (95% CI = 6.88, 15.13), 10.40 (95% CI = 7.13, 14.38), and 2.48 (95% CI = 1.88, 3.88), respectively (Figure 1.3). Mean occupancy estimates for the mature forest, young forest, and habitat generalist guilds were 0.36 (95% CI = 0.15, 0.62), 0.69 (95% CI = 0.37, 0.92), and 0.38 (95% CI = 0.05, 0.86), respectively (Figure 1.4). Estimated mean occupancy of the mature forest guild was negatively associated with OFRA sites, whereas occupancy of the young forest guild was positively associated with OFRA sites (Figure 1.5). Estimated mean occupancy of the habitat generalist guild did not show a meaningful association with OFRA sites. Of the 15 avian indicator species of RS-NH forests detected in our study, occupancy estimates for five species were negatively associated with OFRA sites, and three species were positively associated with OFRA sites (Table 1.2).

1.3.2.3. Naturally Regenerated Forest (>40 yr) Treatment

Mean species richness estimates for the mature forest, young forest, and habitat generalist guilds were 17.73 (95% CI = 12.63, 22.88), 2.25 (95% CI = 1.38, 4.75), and 2.36 (95% CI = 1.75, 3.75), respectively (Figure 1.3). Mean occupancy estimates for the mature forest, young forest, and habitat generalist guilds were 0.74 (95% CI = 0.46, 0.93), 0.10 (95% CI = 0.03, 0.25), and 0.35 (95% CI = 0.04, 0.84), respectively (Figure 1.4). Estimated mean occupancy of the young forest guild was negatively associated with REGEN sites, whereas mean occupancy estimates for both the mature forest and habitat generalist guilds did not show a meaningful association with REGEN sites (Figure 1.5). Of the 15 avian indicator species of RS-NH forests detected in our study, occupancy estimates for two species were negatively associated with REGEN sites, and none were positively associated with REGEN sites (Table 1.2).

1.3.2.4 Mature Forest (>100 yr) Treatment

Mean species richness estimates for the mature forest, young forest, and habitat generalist guilds were 17.48 (95% CI = 12.63, 22.88), 5.50 (95% CI = 3.63, 8.75), and 2.07 (95% CI = 1.50, 3.38), respectively (Figure 1.3). Mean occupancy estimates for the mature forest, young forest, and habitat generalist guilds were 0.71 (95% CI = 0.46, 0.89), 0.34 (95% CI = 0.15, 0.59), and 0.36 (95% CI = 0.07, 0.78), respectively (Figure 1.4). Mean occupancy estimates for the mature forest, young forest, and habitat generalist guilds did not show a meaningful association with MAT sites (Figure 1.5). Of the 15 avian indicator species of RS-NH forests detected in our study, none were negatively associated with MAT sites, and seven species were positively associated with MAT sites (Table 2.2).

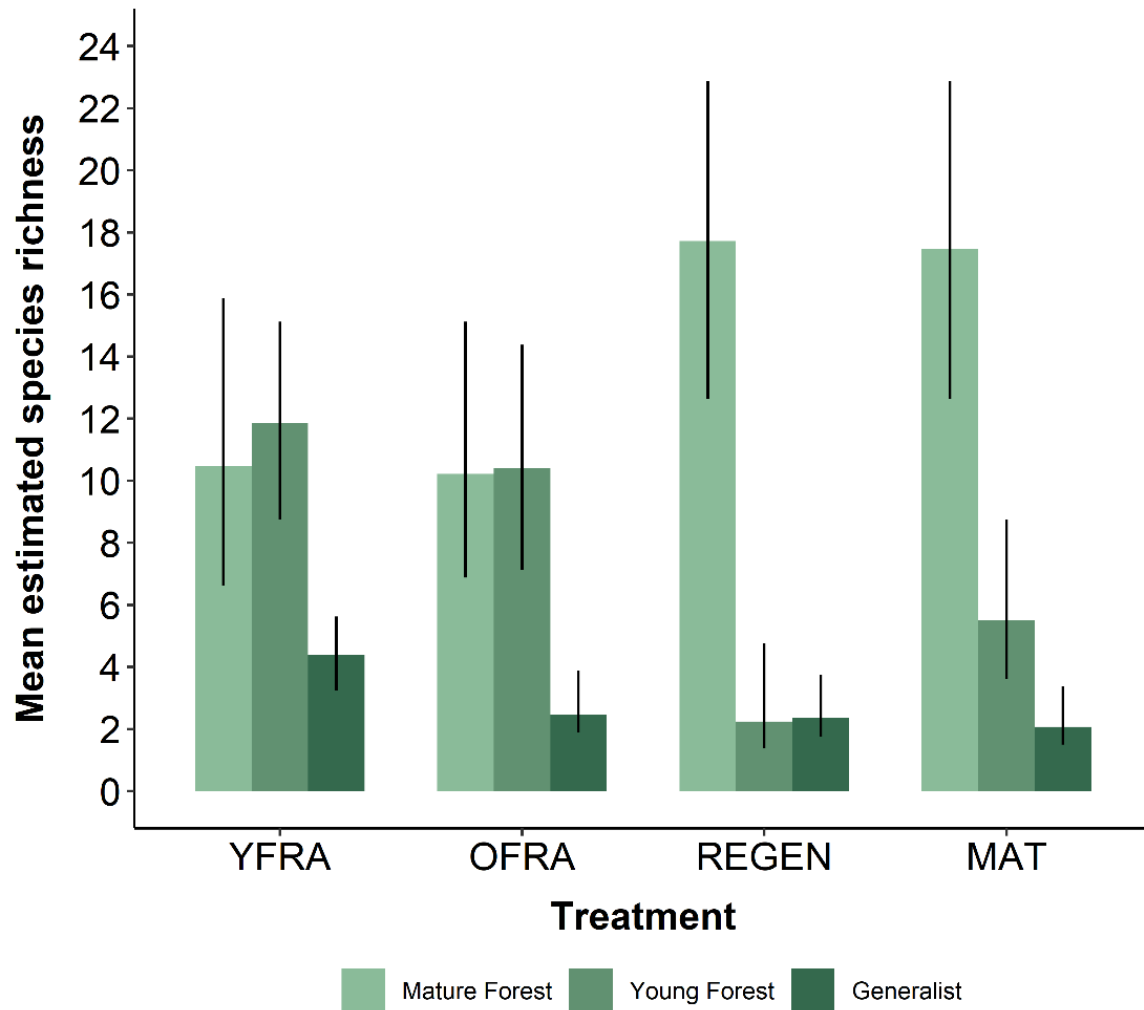


Figure 1.3. Mean estimated species richness of three avian nesting habitat guilds (i.e., mature forest, young forest, generalist) across four treatment types (i.e., younger FRA, older FRA, naturally regenerated forest on mineland, unmined mature forest). Solid black lines indicate 95% credible intervals for richness estimates.

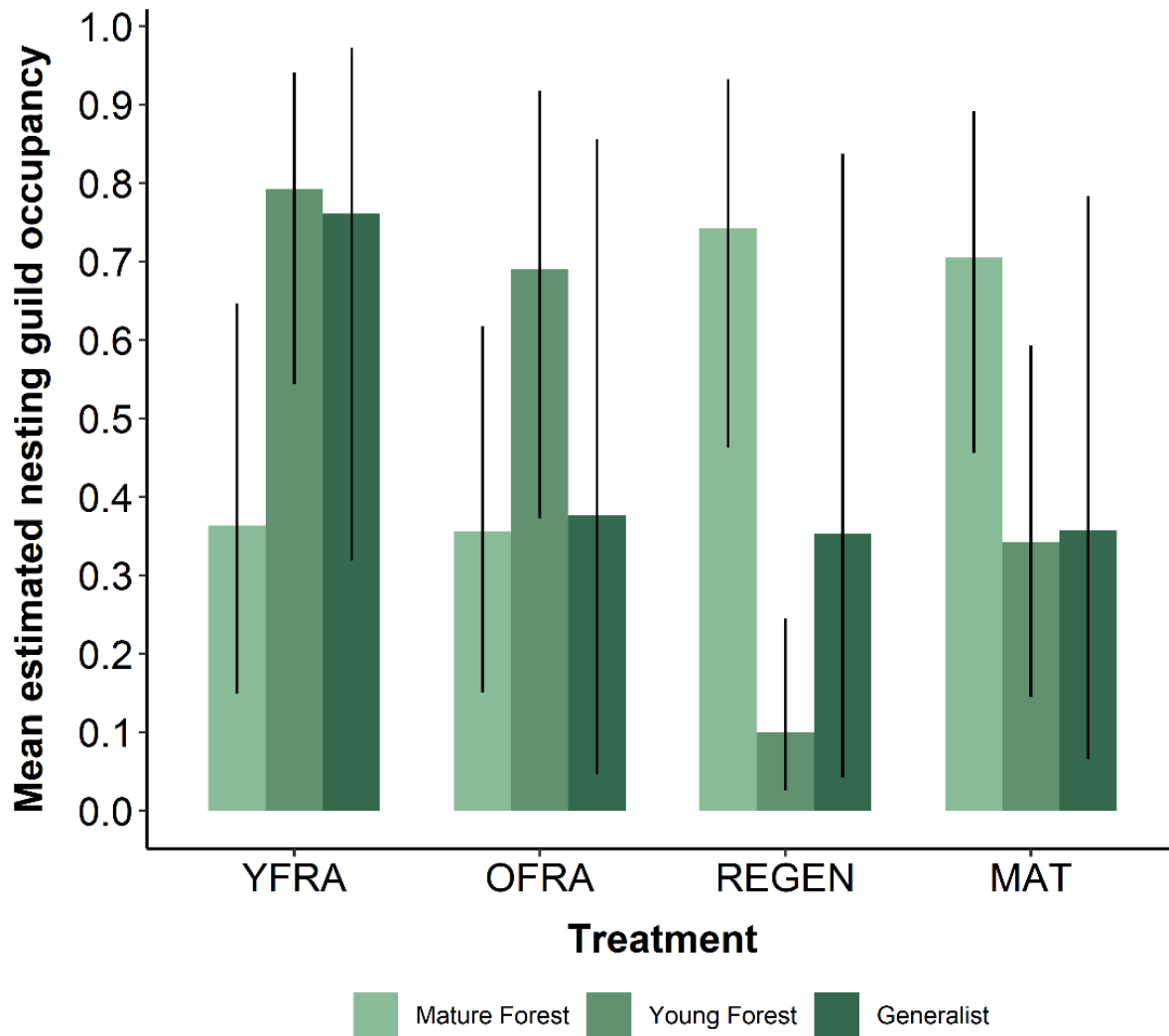


Figure 1.4. Mean estimated occupancy of three avian nesting habitat guilds (i.e., mature forest, young forest, generalist) across four treatment types (i.e., younger FRA, older FRA, naturally regenerated forest on mineland, unmined mature forest). Solid black lines indicate 95% credible intervals for occupancy estimates.

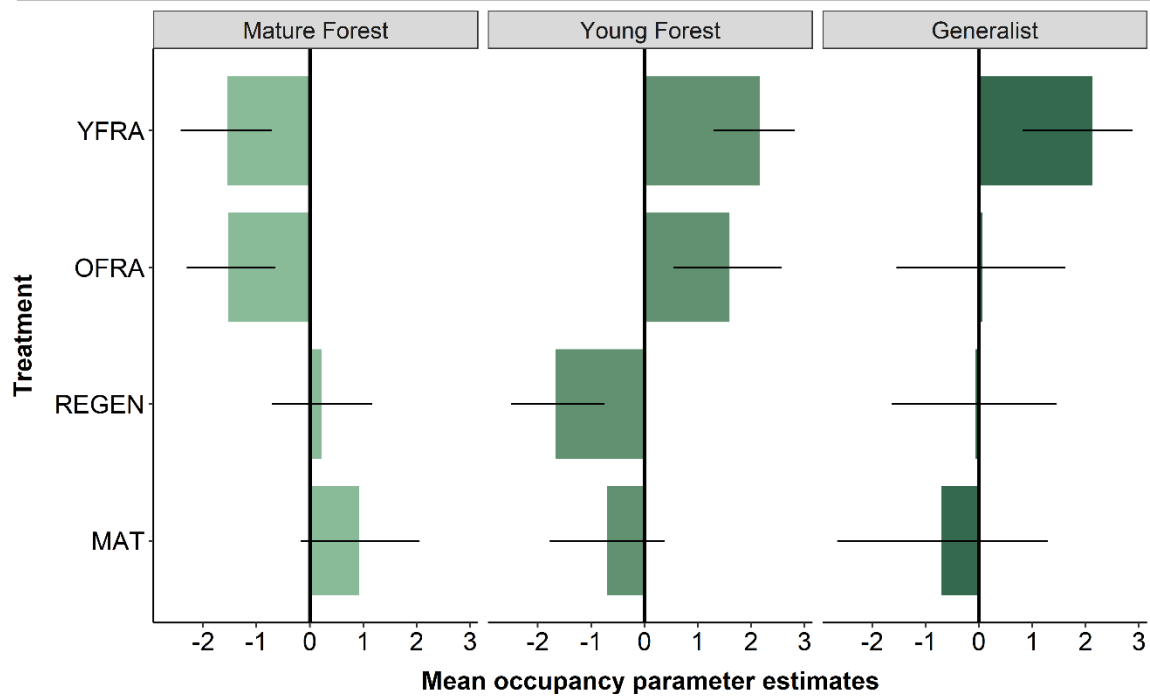


Figure 1.5. Mean parameter estimates for occupancy of three avian nesting habitat guilds (i.e., mature forest, young forest, generalist). Green bars to the right of 0.0 indicate a positive relationship between guild occupancy and treatment, while bars to the left of 0.0 indicate a negative relationship. Solid black lines indicate 95% credible intervals (95% CIs) for parameter estimates. Estimates are considered meaningful if the 95% CIs do not contain zero.

Table 1.2. Mean parameter estimates (α) and 95% credible intervals (95% CIs) for occupancy of RS-NH forest avian indicator species, separated by avian nesting habitat guild (i.e., mature forest, young forest, generalist).

Guild	Species	α_{YFRA} (95% CI)	α_{OFRA} (95% CI)	α_{REGEN} (95% CI)	α_{MAT} (95% CI)
Mature Forest	Blackburnian Warbler	-1.86 (-2.85, -0.82)	-1.68 (-2.68, -0.68)	0.54 (-0.98, 2.36)	2.16 (-0.04, 4.55)
	Black-throated Green Warbler	-1.57 (-2.60, -0.54)	-1.40 (-2.53, 0.32)	0.45 (-1.26, 2.36)	4.12 (1.66, 6.76)
	Blue-headed Vireo	-1.61 (-2.64, -0.58)	-1.57 (-2.62, -0.52)	0.35 (-1.36, 2.28)	3.08 (0.74, 5.79)
	Golden-crowned Kinglet	-1.52 (-2.53, -0.51)	-1.91 (-2.85, -0.97)	0.89 (-0.59, 2.66)	2.21 (0.00, 4.57)
	Hermit Thrush	-1.69 (-2.74, -0.64)	-1.35 (-2.50, 0.42)	0.22 (-1.62, 2.18)	3.21 (0.72, 6.22)
	Magnolia Warbler	-1.63 (-2.66, -0.60)	-1.36 (-2.50, 0.41)	0.30 (-1.52, 2.26)	4.01 (1.57, 6.69)
	Red-breasted Nuthatch	-1.59 (-2.65, -0.53)	-1.31 (-2.50, 0.60)	0.34 (-1.47, 2.29)	3.51 (0.76, 6.64)
	Rose-breasted Grosbeak	-1.56 (-2.65, -0.47)	-1.60 (-2.70, -0.50)	0.38 (-1.43, 2.30)	0.37 (-3.45, 5.05)
	Swainson's Thrush	-1.39 (-2.43, -0.35)	-2.06 (-2.91, -1.21)	0.49 (-1.03, 2.30)	2.20 (-0.01, 4.53)
	Winter Wren	-1.68 (-2.74, -0.62)	-1.40 (-2.55, 0.31)	0.41 (-1.27, 2.29)	1.75 (-1.10, 5.32)
	Yellow-rumped Warbler	-1.37 (-2.45, 0.19)	-1.47 (-2.58, 0.25)	0.11 (-1.90, 2.14)	3.78 (1.13, 6.70)
Young Forest	Chestnut-sided Warbler	2.15 (1.03, 3.27)	1.77 (0.46, 3.08)	-1.74 (-2.78, -0.70)	-1.72 (-4.00, 0.61)
	Dark-eyed Junco	2.17 (1.01, 3.33)	1.69 (0.16, 3.22)	-1.37 (-2.55, 0.32)	1.11 (-1.32, 3.67)
	Mourning Warbler	2.00 (0.31, 3.69)	1.92 (0.67, 3.17)	-1.66 (-2.76, -0.56)	-2.54 (-5.65, 1.09)
Generalist	Northern Flicker	2.06 (0.33, 3.79)	0.52 (-1.36, 2.54)	-0.04 (-1.99, 1.89)	-2.06 (-6.08, 3.63)

*Alpha (α) values indicate the strength and direction of the relationship between species occupancy and treatment type (i.e., younger FRA (YFRA; 2-5 yr), older FRA (OFRA; 8-11 yr), naturally regenerated forest on mineland (REGEN; >40 yr), and unmined mature forest (MAT; >100 yr)). Parameter estimates in boldface indicate meaningful relationships between species occupancy and the given treatment type (i.e., 95% CIs did not contain zero).

1.3.2.5 Abundance Estimates

1.3.2.5.1. Mature Forest Guild

Abundance estimates of mature forest guild species varied across treatment types. Species with the greatest estimated abundances in the MAT treatment included Black-throated Green Warblers ($\beta_1 = 1.712$; $sd = 0.521$; 95% CI = 0.911, 3.003) (Figure 1.6) and Magnolia Warblers ($\beta_1 = 1.991$; $sd = 0.593$; 95% CI = 1.079, 3.499) (Figure 1.7), while Golden-crowned Kinglets were most abundant in the REGEN treatment ($\beta_4 = 1.342$; $sd = 0.591$; 95% CI = 0.224, 2.509) (Figure 1.8). Conversely, the OFRA treatment was negatively associated with estimated abundances of Golden-crowned Kinglets ($\beta_3 = -2.550$; $sd = 1.434$; 95% CI = -5.757, -0.143) (Figure 1.8) and Swainson's Thrushes ($\beta_3 = -1.902$; $sd = 0.960$; 95% CI = -4.034, -0.209) (Figure 1.9). Furthermore, Magnolia Warblers were least abundant in YFRA sites ($\beta_2 = -1.483$; $sd = 0.485$; 95% CI = -2.525, -0.611) (Figure 1.7). Sampling covariates did not have an influence on the estimated per-individual detection probabilities of any mature forest guild species.

1.3.2.5.2. Disturbed (Shrubland/Forest Edge) Guild

The estimated abundance of disturbed habitat species varied across treatment types. Species with abundance

estimates positively associated with the YFRA treatment included the Eastern Towhee ($\beta_2 = 3.417$; $sd = 1.216$; 95% CI = 1.559, 6.276) (Figure), Field Sparrow ($\beta_2 = 2.735$; $sd = 0.852$; 95% CI = 1.263, 4.601) (Figure), Indigo Bunting ($\beta_2 = 1.596$; $sd = 0.603$; 95% CI = 0.507, 2.882) (Figure), and Song Sparrow ($\beta_2 = 6.819$; $sd = 2.801$; 95% CI = 2.769, 13.390) (Figure 1.13). Similarly, the OFRA treatment was positively associated with abundance estimates of the Eastern Towhee ($\beta_3 = 3.955$; $sd = 1.233$; 95% CI = 2.052, 6.840) (Figure 1.10), Field Sparrow ($\beta_3 = 2.779$; $sd = 0.898$; 95% CI = 1.211, 4.741) (Figure), Indigo Bunting ($\beta_3 = 1.656$; $sd = 0.637$; 95% CI = 0.489, 2.987) (Figure), and Song Sparrow ($\beta_3 = 5.172$; $sd = 2.833$; 95% CI = 0.948, 11.760) (Figure 1.13). Dark-eyed Junco abundance was positively associated with the OFRA ($\beta_3 = 0.829$; $sd = 0.349$; 95% CI = 0.143, 1.515) and MAT treatments ($\beta_1 = 1.423$; $sd = 0.394$; 95% CI = 0.748, 2.345) (Figure). Conversely, the MAT treatment was negatively associated with estimated abundances of Eastern Towhees ($\beta_1 = -2.013$; $sd = 1.212$; 95% CI = -4.867, -0.059) and Song Sparrows ($\beta_1 = -5.218$; $sd = 2.801$; 95% CI = -11.841, -1.155). Additionally, Field Sparrows ($\beta_4 = -8.449$; $sd = 5.979$; 95% CI = -22.891, -0.389) and Indigo Buntings ($\beta_4 = -9.124$; $sd = 5.794$; 95% CI = -22.680, -1.333) were least abundant across REGEN sites. Some estimated per-individual detection probabilities were influenced by sampling parameters. Eastern Towhees were detected later in the day ($\alpha_3 = 0.551$; $sd = 0.242$; 95% CI = 0.103, 1.061), while Field Sparrows were detected less as air temperature increased ($\alpha_6 = -0.691$; $sd = 0.346$; 95% CI = -1.420, -0.071). Indigo Bunting estimated detection probability was negatively associated with day of year ($\alpha_2 = -0.772$; $sd = 0.283$; 95% CI = -1.350, -0.250) and cloud cover ($\alpha_4 = -0.628$; $sd = 0.273$; 95% CI = -1.184, -0.122).

1.3.2.5.3. Habitat Generalist Guild

We performed abundance analyses on three species from the habitat generalist guild. Blue Jays were most abundant in the MAT treatment ($\beta_1 = 1.320$; $sd = 0.770$; 95% CI = 0.103, 3.297) (Figure), while American Robins had the highest abundance in both the MAT ($\beta_1 = 1.604$; $sd = 0.595$; 95% CI = 0.713, 3.222) and YFRA treatments ($\beta_2 = 0.903$; $sd = 0.325$; 95% CI = 0.275, 1.544) (Figure). Estimated abundances of American Robins was also negatively associated with increasing distance from the nearest road ($\beta_5 = -0.352$; $sd = 0.138$; 95% CI = -0.625, -0.089) (Figure 1.16). Treatment type did not have an influence on the estimated abundance of American Crows; however, their per-individual detection probabilities were negatively associated with day of year ($\alpha_2 = -0.638$; $sd = 0.319$; 95% CI = -1.306, -0.057).

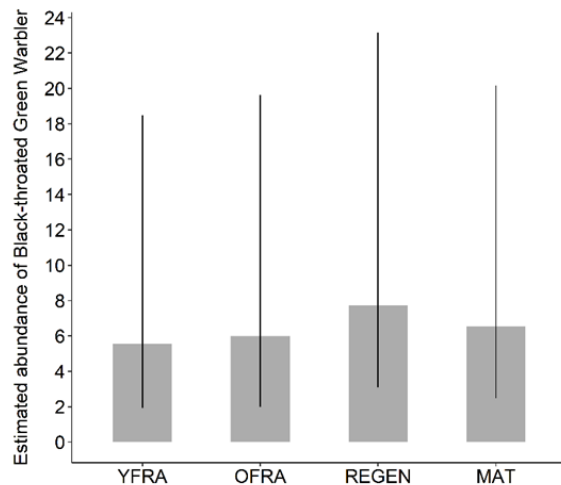


Figure 1.6. Estimated mean abundance of the Black-throated Green Warbler (*Setophaga virens*) across treatment types. $\Psi_{YFRA} = 5.564$ (95% CI = 1.924, 18.482), $\Psi_{OFRA} = 5.990$ (95% CI = 1.984, 19.629), $\Psi_{REGEN} = 7.722$ (95% CI = 3.101, 23.148), and $\Psi_{MAT} = 6.551$ (95% CI = 2.488, 20.148).

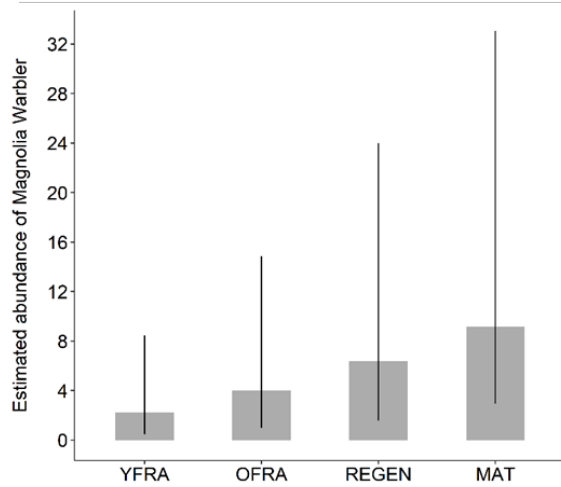


Figure 1.7. Estimated mean abundance of the Magnolia Warbler (*Setophaga magnolia*) across treatment types. $\Psi_{YFRA} = 2.223$ (95% CI = 0.480, 8.464), $\Psi_{OFRA} = 4.020$ (95% CI = 1.014, 14.886), $\Psi_{REGEN} = 6.378$ (95% CI = 1.572, 23.991), and $\Psi_{MAT} = 9.167$ (95% CI = 2.942, 33.084).

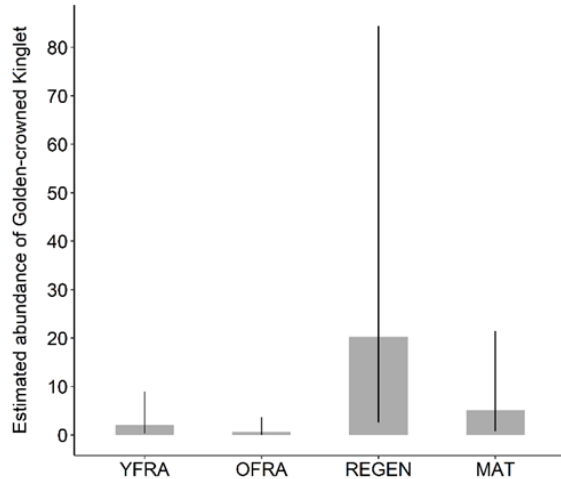


Figure 1.8. Estimated mean abundance of the Golden-crowned Kinglet (*Regulus satrapa*) across treatment types. $\Psi_{YFRA} = 2.155$ (95% CI = 0.311, 8.967), $\Psi_{OFRA} = 0.702$ (95% CI = 0.010, 3.702), $\Psi_{REGEN} = 20.260$ (95% CI = 2.627, 84.456), and $\Psi_{MAT} = 5.131$ (95% CI = 0.744, 21.457).

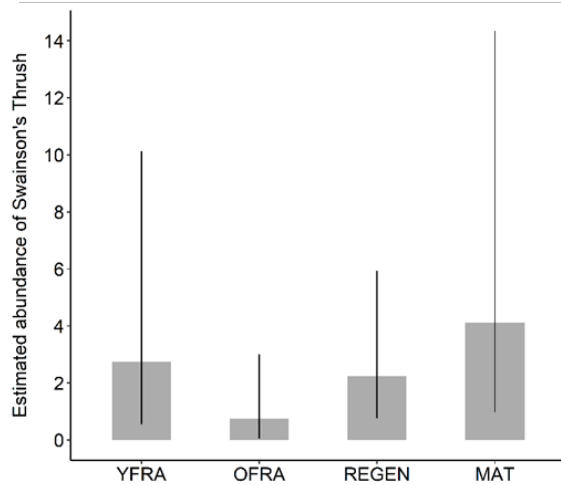


Figure 1.9. Estimated mean abundance of the Swainson's Thrush (*Catharus ustulatus*) across treatment types. $\Psi_{YFRA} = 2.753$ (95% CI = 0.546, 10.133), $\Psi_{OFRA} = 0.753$ (95% CI = 0.059, 3.002), $\Psi_{REGEN} = 2.247$ (95% CI = 0.762, 5.942), and $\Psi_{MAT} = 4.108$ (95% CI = 0.989, 14.355).

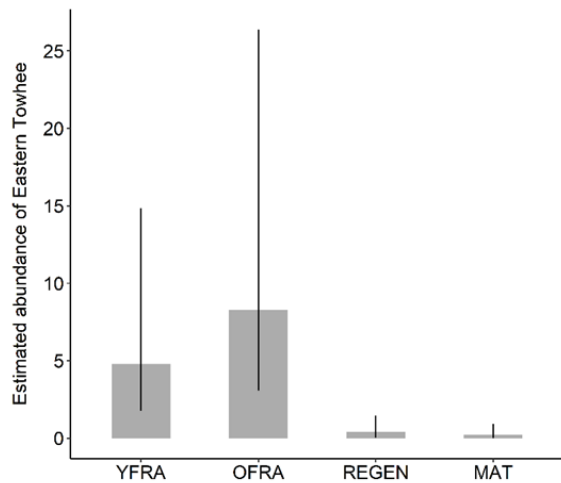


Figure 1.10. Estimated mean abundance of the Eastern Towhee (*Pipilo erythrophthalmus*) across treatment types. $\Psi_{YFRA} = 4.788$ (95% CI = 1.786, 14.826), $\Psi_{OFRA} = 8.262$ (95% CI = 3.061, 26.372), $\Psi_{REGEN} = 0.398$ (95% CI = 0.041, 1.459), and $\Psi_{MAT} = 0.236$ (95% CI = 0.008, 0.943).

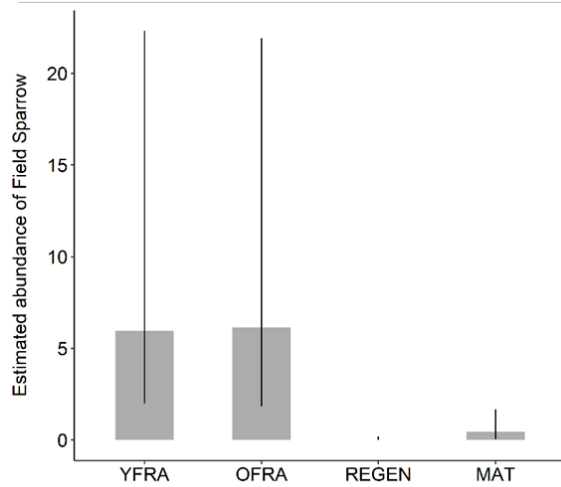


Figure 1.11. Estimated mean abundance of the Field Sparrow (*Spizella pusilla*) across treatment types. $\Psi_{YFRA} = 5.953$ (95% CI = 1.988, 22.327), $\Psi_{OFRA} = 6.143$ (95% CI = 1.839, 21.924), $\Psi_{REGEN} = 0.020$ (95% CI = 0.000, 0.187), and $\Psi_{MAT} = 0.447$ (95% CI = 0.047, 1.664).

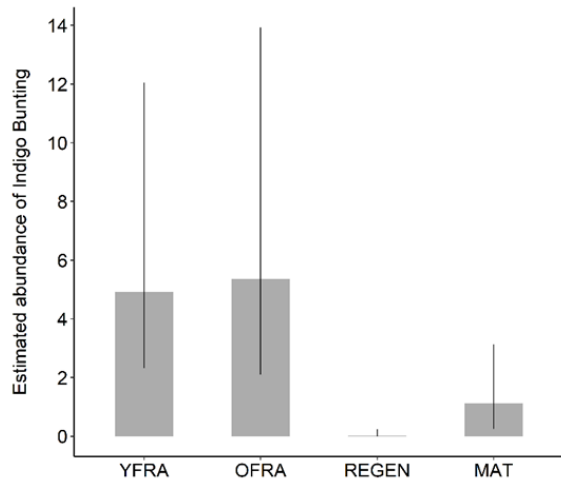


Figure 1.12. Estimated mean abundance of the Indigo Bunting (*Passerina cyanea*) across treatment types. $\Psi_{YFRA} = 4.918$ (95% CI = 2.318, 12.055), $\Psi_{OFRA} = 5.366$ (95% CI = 2.101, 13.923), $\Psi_{REGEN} = 0.025$ (95% CI = 0.000, 0.236), and $\Psi_{MAT} = 1.113$ (95% CI = 0.248, 3.133).

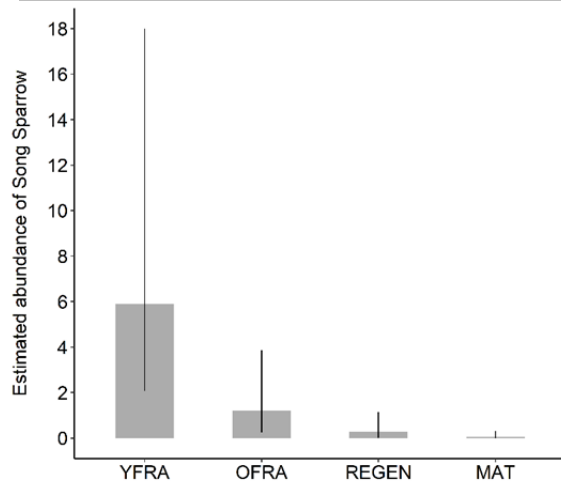


Figure 1.13. Estimated mean abundance of the Song Sparrow (*Melospiza melodia*) across treatment types. $\Psi_{YFRA} = 5.894$ (95% CI = 2.075, 17.994), $\Psi_{OFRA} = 1.218$ (95% CI = 0.256, 3.865), $\Psi_{REGEN} = 0.269$ (95% CI = 0.005, 1.142), and $\Psi_{MAT} = 0.045$ (95% CI = 0.000, 0.315).

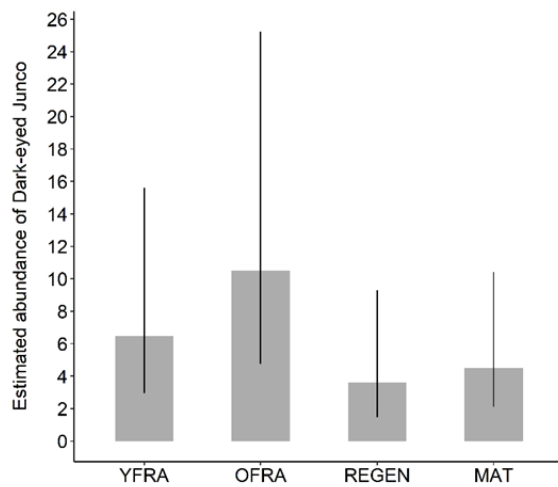


Figure 1.14. Estimated mean abundance of the Dark-eyed Junco (*Junco hyemalis*) across treatment types. $\Psi_{YFRA} = 6.460$ (95% CI = 2.955, 15.612), $\Psi_{OFRA} = 10.495$ (95% CI = 4.751, 25.231), $\Psi_{REGEN} = 3.601$ (95% CI = 1.478, 9.274), and $\Psi_{MAT} = 4.518$ (95% CI = 2.113, 10.434).

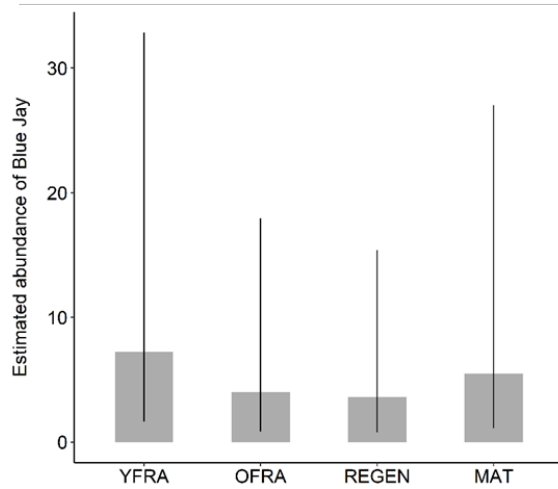


Figure 1.15. Estimated mean abundance of the Blue Jay (*Cyanocitta cristata*) across treatment types. $\Psi_{YFRA} = 7.228$ (95% CI = 1.655, 32.836), $\Psi_{OFRA} = 3.998$ (95% CI = 0.857, 17.965), $\Psi_{REGEN} = 3.595$ (95% CI = 0.768, 15.405), and $\Psi_{MAT} = 5.499$ (95% CI = 1.109, 27.033).

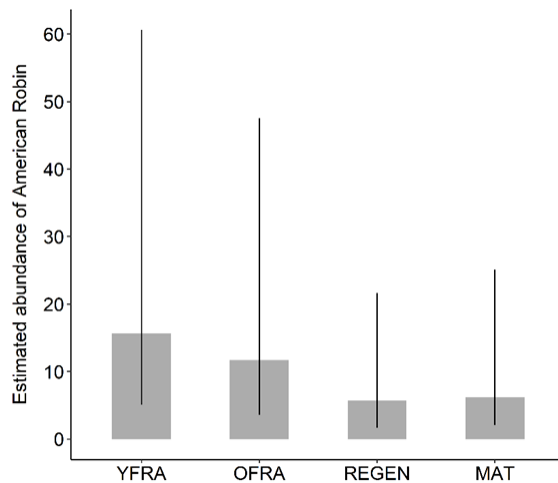


Figure 1.16. Estimated mean abundance of the American Robin (*Turdus migratorius*) across treatment types. $\Psi_{YFRA} = 15.614$ (95% CI = 5.089, 60.653), $\Psi_{OFRA} = 11.692$ (95% CI = 3.557, 47.550), $\Psi_{REGEN} = 5.742$ (95% CI = 1.716, 21.630), and $\Psi_{MAT} = 6.231$ (95% CI = 2.039, 25.083).

1.4. Discussion

One of the primary objectives of our study was to compare the composition of breeding bird communities among FRA forests, naturally regenerated forests on mineland, and unmined mature forests. In our study, forests established using the FRA were dominated by avian species that breed in young forests, while naturally regenerated forests on mineland and mature forests on unmined sites supported higher proportions of mature forest bird species. This finding did not align with our prediction that accelerated succession in 11-yr old forests established using the FRA and arrested succession in >40-yr old naturally regenerated forests would result in conditions that supported similar avian communities. Although forest succession on minelands reclaimed using the FRA is likely faster compared to a more passive approach through natural regeneration (Groninger et al. 2007), our analyses demonstrate that forests established using the FRA require more time to provide habitat for mature forest breeding birds. Indeed, a previous study in eastern forests found that even-aged forests stands promoted the diversity of mature forest bird species in as few as 20 years post-timber harvest (McDermott et al. 2011; Duguid et al. 2016). Continued monitoring of our sites will

help characterize the general timing of when minelands reforested using the FRA transition to hosting mature forest bird communities.

Patterns in breeding bird occupancy followed similar trends to species richness, where FRA forests had higher occupancy by the young forest guild and lower occupancy by the mature forest guild. We also found that young forest breeding birds were least likely to occur in naturally regenerated forests. This finding offers evidence that within one decade, FRA forests are likely providing more habitat for young forest breeding birds compared to non-FRA reclaimed forests that have passively regenerated for 3-4 additional decades. In order to maintain herbaceous, shrub, and sampling cover for birds associated with young forest habitat, land managers may consider staggering implementation of the FRA across legacy mines so that young forests are continuously established proximate to mature forests. Dynamic forest structures consisting of multi-aged forest patches will sustain the habitat needs of declining young forest bird communities (Sauer et al. 2017). Additionally, several mature forest bird species have been documented using young forests during post-fledgling periods, as long as such forest is proximate to their mature forest nesting habitat (Anders, Faaborg, and Thompson 1998; Fiss et al. 2021; King et al. 2006; Raybuck et al. 2022; Vitz and Rodewald 2006). Therefore, understanding the timing of FRA forest succession may allow land managers to implement the FRA in a way that satisfies the spatiotemporal breeding habitat use patterns of forest bird species.

Our second research objective was to assess the occurrence of RS-NH forest avian indicator species within FRA forest and naturally regenerated forest on mineland compared to unmined mature forest. We found that more RS-NH forest indicator species were positively associated with unmined mature forest reference sites compared to forest established using the FRA; however, this is unsurprising considering that 11 of the 15 detected indicator species belonged to the mature forest nesting guild. Most importantly, both younger and older FRA treatments were positively associated with the occupancy of all three RS-NH indicator species (i.e., Chestnut-sided Warbler [*Setophaga pensylvanica*], Dark-eyed Junco [*Junco hyemalis*], and Mourning Warbler [*Geothlypis philadelphia*]) that nest in young forests. This finding suggests that the FRA forests we monitored have already advanced to a successional stage that is known to be used by species associated with young RS-NH forest. Furthermore, we found that the number of mature forest indicator species negatively associated with FRA forests decreased from 10 to 5 between the younger and older FRA treatments. Collectively, these findings suggest that breeding bird communities occupying FRA sites include species indicative of RS-NH reference systems, and that given additional time to successional advance, FRA forests will likely support more of the bird species that are indicators of mature RS-NH forest.

Given the older age of naturally regenerated forests on mineland (i.e., >40 yrs), we expected that this treatment may foreshadow habitat availability within FRA forests for breeding birds. However, none of the avian species that are indicators of RS-NH forests were positively associated with naturally regenerated forests. In this regard, naturally regenerated forests on mineland substantially differed from forests established using the FRA and unmined mature forests, both of which supported the indicator species corresponding to their successional stage. Even though naturally regenerated forests and unmined forests were estimated to host a similar overall richness of mature forest species (i.e., ~18 species, Figure 3), naturally regenerated forests are not providing conditions that support bird communities expected within the pre-mining RS-NH forest type (Clipp et al. 2022). The presence of mature forest species yet dearth of RS-NH indicator species within forests that naturally regenerated on mineland may point to limitations of RS-NH recolonization via passive seed recruitment (Rhodes 2022).

Associations with RS-NH indicator species occupancy suggest that forests established using the FRA may ultimately surpass naturally regenerated forests in providing habitat for RS-NH-dependent bird species. This is likely because forests resulting from the FRA were intentionally managed for growing conditions that facilitate RS-NH forest establishment and growth, while growth of naturally regenerated forests on previously mined lands was stifled by heavy soil compaction and exotic species planting (Green Forests Work 2020). Even so, the narrow mine benches that characterized our naturally regenerated sites exhibited a higher likelihood of naturally regenerating into native forest compared to large surface mines reclaimed under SMCRA legislation, an observation made elsewhere in the Appalachians (Angel et al. 2015; Green Forests Work 2020). Mountaintop removal mines and expansive area mines that are heavily compacted and converted to homogenous grassland or shrubland often remain in a state of arrested succession for several decades (Ingold and Dooley 2013; Margenau et al. 2019; Wood and Ammer 2015). Not only could the FRA become a viable alternative to traditional reclamation that establishes grassland on active mines, but it could be more widely

used to convert legacy mines into native forests (Burger et al. 2013). Future studies that periodically assess wildlife communities on our study sites will be valuable for adaptively modifying FRA protocols to best achieve desired reclamation outcomes.

1.5. Conclusion

In order to evaluate the efficacy of the FRA for Appalachian forest bird communities, we compared the species richness and occupancy of avian nesting habitat guilds, and more specifically, RS-NH avian indicator species, between FRA forests and nearby reference forests. This study was the first to empirically assess avian community response to the FRA. Our preliminary assessment suggests that at present, legacy mines reclaimed using the FRA are providing habitat for birds that breed in young forests. In particular, the FRA avian community assemblage included all three RS-NH avian indicator species associated with young RS-NH forests. While forests established using the FRA were still largely unoccupied by mature forest birds at 11-yr post-reclamation, occupancy of avian species that are indicators of mature RS-NH forests increased with FRA forest age. This suggests that continued succession of FRA forests may improve habitat availability for diverse breeding bird communities typical of RS-NH forests. As forest-obligate Appalachian birds continue to face steep declines (Franzreb and Phillips 1996; Franzreb and Rosenberg 1997), the prospect of habitat recovery has become increasingly urgent. Our research indicated that implementing the FRA on active and legacy mines may help to restore native forest cover that can meet the habitat needs of diverse forest bird communities.

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Section 2. Bat Activity on High Elevation Reforested Coal Mines in the Monongahela National Forest, West Virginia

2.1. Introduction

The Forestry Reclamation Approach (FRA), introduced in 2005, was created to combat the loss of 600,000 ha of forest habitat in the Appalachian region due to surface coal mining (Zipper et al. 2011, Barton et al. 2018). Under conventional reclamation practices, created grasslands/shrublands can remain in a state of arrested succession that may endure for decades if not centuries (Angel et al. 2005). To reforest active and abandoned mined lands, the FRA directs the creation of a loosely graded soil medium and the planting of appropriate ground cover and trees (Zipper et al. 2011). Where suitable, depressional wetlands can be created as an additional step in the FRA restoration process. Wetlands provide numerous services including nutrient cycling, flood regulation, water storage, water treatment, and habitat for many aquatic and terrestrial organisms (Shartz 2014). Though wetlands are an important feature of the Appalachian landscape (Calhoun et al. 2012), the purposeful creation of wetlands for wildlife habitat on reclaimed mined lands is not common as most created water features are required to be removed during the reclamation process (P. Angel, Office of Surface Mining, pers. comm.). Creating wetlands as part of the FRA process restores some of this lost habitat. Mined lands restored using the FRA can have improved hydrology, water quality, soil chemistry, tree growth, and tree survival relative to conventionally reclaimed mined land (Sena et al. 2014, Miller et al. 2015, Agouridis et al. 2018, Barton et al. 2018, Dement et al. 2020, Williamson and Barton 2020). Studies of wildlife on mined lands have primarily focused on lands reclaimed to grasslands or shrublands (Lituma et al. 2021) not only because of their prevalence, but also because forests take decades to grow and mature. Consequently, only a few studies have researched wildlife use of mined land restored via the FRA. At one restored legacy mined land in West Virginia, created vernal pools were shown to support up to eight species of amphibians (Lambert et al. 2021). Using herbivore exclusion techniques on reforested mined land, Hackworth et al. (2018) confirmed that *Odocoileus virginianus* (white-tailed deer), *Cervus canadensis* (elk), *Sylvilagus* spp. (rabbits), and small mammals browse planted trees. Lastly, following the FRA produces a heterogeneous ground surface of vegetative debris, rocks, and holes, which results in increased small mammal biodiversity compared to other mined land treatments (Larkin et al. 2008). While these studies provide evidence that FRA-restored mined lands can benefit native fauna, further research is needed.

One group of species that could benefit from restoration efforts on surface mines is bats. There are 25 species of bats in North America that use forests for roosting and/or foraging (Kurta et al. 2007). Because forests are integral habitat for bats, the loss and fragmentation of forests is a contributing factor in bat population declines (Frick et al. 2019). Another major factor is White-nose Syndrome (WNS), an infectious disease caused by the pathogenic fungus *Pseudogymnoascus destructans*. The fungus causes bats to arouse frequently during their hibernation and use their fat stores before spring (Frick et al. 2016). Habitat loss and WNS largely contributed to the listing of four bat species from the Appalachian Mountains to the Federal Endangered Species List: *Myotis grisescens* (gray bat), *M. septentrionalis* (northern long-eared bat), *M. sodalis* (Indiana bat), and *Corynorhinus townsendii virginianus* (Virginia big-eared bat) (USFWS 1967, 1976, 1979, and 2022). Cheng et al. (2019) presented evidence that increased fat storage when entering hibernation helps reduce the mortality rate of WNS, hence increasing roosting and foraging habitat may benefit these and other bat species in Appalachia. FRA-restored areas eventually grow into mature forest, which forest bats need for roosting. However, bats can forage for insects in and above forests, along forest edges, above bodies of water, and in open areas like fields or recent timber harvests (Brigham 2007, Lacki et al. 2007, Loeb and O'Keefe 2011). The initial years of mined land restoration have forest edges and open patches, but these potential foraging spaces will mature to interior forest. Although bats utilize interior forest as foraging habitat, they have greater foraging activity at bodies of water (Zimmerman and Glanz 2000, Owen et al. 2004, Menzel et al. 2005). In particular, wetlands are ideal foraging locations. Since bats rely on echolocation to avoid objects in flight as well as to home-in on insect prey, bats tend to forage more in habitats with fewer physical obstructions, or clutter (Fenton 1990), in their airspace to make hunting less energetically expensive

A modified version of this section is currently published in Ecological Restoration.. Snyder, B., C. Barton, M. Lacki, S.J. Price, and Z. Hackworth. 2024. Bat Activity on High elevation reforested coal mines in the Monongahela National Forest, West Virginia. Ecological Restoration 42:108-122.

(Zimmerman and Glanz 2000, Owen et al. 2004, Moore and Best 2018). Wetlands can provide less cluttered airspace, particularly if they have open canopies. Bats also tend to forage more at still water because running water may create excess ultrasonic noise, which makes it harder for bats to hunt (Frenckell and Barclay 1987, Mackey and Barclay 1989). Bats also require open water sources for drinking (Russo et al. 2012). The ability to eat and drink at the same location reduces commuting distances and further lowers energy costs. By including these foraging areas, reforestation coupled with wetland creation could produce better bat habitat than reforestation alone.

The FRA is relatively untested for its impacts on terrestrial wildlife. While the establishment of a plant community is a crucial step, ecological restoration is not complete until the land recruits and retains appropriate wildlife species. Since bats are one focal group that may benefit from restored mined land, we aimed to examine whether 1) bats use FRA-restored lands as foraging habitat, 2) bat activity at wetlands in FRA-restored lands is comparable to that at wetlands in traditionally reclaimed mined land and mature forest, and 3) management or restoration techniques can be implemented to foster bat activity in FRA-restored lands.

2.2. Methods

2.2.1. Study Area

The study location was Cheat Mountain (38°33'11.71"N, 79°56'22.99"W) within the Monongahela National Forest (MNF) in east-central West Virginia (WV) (Figure 2.1). Most of the study area was located within the Mower Tract, a 16,000 ha parcel of land that was logged during the industrial logging era of the late 1800s and early 1900s. Approximately 600 ha of the tract were also surface mined for coal in the 1970's before the entire parcel became part of the MNF in the mid-1980s. Hardwood forests have grown to maturity across the Mower Tract, except in areas that were surfaced mined. The mined areas had been reclaimed with non-native grasses and conifers and entered a state of arrested succession due to compacted soils and competition from planted vegetation (Branduzzi et al. 2023). In 2010, 36 ha of the previously reclaimed land were restored to native forest using the FRA. During the restoration process, wetlands were also created to intercept and retain precipitation, catch sediment, and provide wildlife habitat (U.S.D.A. Forest Service 2016). Restoration of additional mined land within the Mower Tract has occurred annually (except for 2012). Site preparation activities start by knocking down non-native trees and piling the brush. Bulldozers equipped with dual, rear-mounted ripping shanks then cross-rip (deep plow) the site before the felled trees are scattered across the restoration area with an excavator. Wetlands are created roughly based on the methods outlined by Biebighauser (2003). After evaluating a site for its hydrological properties, a contractor uses an excavator to dig depressional wetlands usually with an area of 5–60 m² to depths of 40–60 cm in areas where seepage or clayey/low permeability soils are present. The interior of the depression is compacted to promote ponding. Woody debris from the tree felling and large rocks unearthed during the ripping are placed in the wetlands to generate microhabitat features. The final step of the restoration process establishes diverse, native vegetation across the site by direct seeding and/or planting native seedlings.

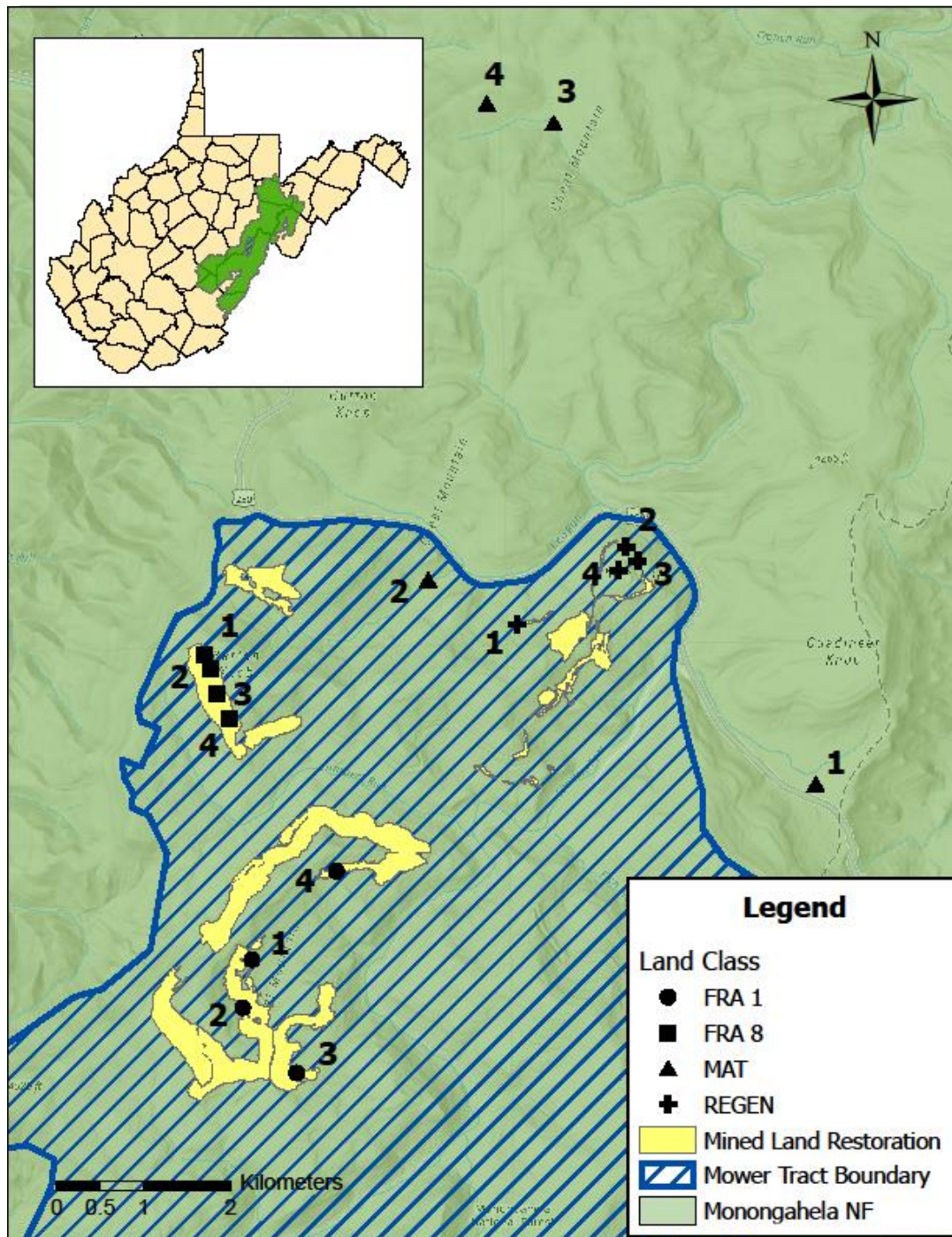


Figure 2.1. Map of the study area. The inset map denotes the Monongahela National Forest in east-central West Virginia. The orange region is the Mower Tract, a parcel of land that was surface mined for coal in the 1970s. The main map shows the survey sites in relation to the Mower Tract. Areas in yellow have been restored using the FRA over the past 12 years. This study examined four land class categories: 1– year-old FRA legacy mined land (FRA1), 8-year-old FRA legacy mined land (FRA8), conventionally reclaimed mined land with naturally regenerating forest (REGEN), and mature forest not previously mined (MAT).

2.2.2. Site Selection

Our study sites included four different land class categories: 1-year-old reforested FRA legacy mined land (FRA1), 8-year-old reforested FRA legacy mined land (FRA8), conventionally reclaimed mined land with

naturally regenerating forest (REGEN), and mature forest not previously mined (MAT).

Site preparation at FRA1 occurred in 2019, and the site was planted in 2020 with a seedling mix of 12 hardwood species, eight shrub species, and *Picea rubens* (red spruce). Trees were planted on 1-m centers. Herbaceous species germinated from the seed bank and covered much of the area. Most planted vegetation did not exceed 1 m in height. FRA8 was restored in 2013–2014 and planted with four hardwood species, two shrub species, and *P. rubens*. The trees and shrubs were approximately 2–4 m in height and herbaceous cover was present. REGEN was mined land reclaimed to grassland approximately 40 years ago. It was a strip of grassland with naturally formed wetlands, and *P. rubens* had begun to recolonize the area. It had mature red spruce-northern hardwood forests outside its boundaries creating a matrix of early successional forest and mature forest. MAT wetlands formed naturally and were surrounded by mature red spruce-northern hardwood forest that was not being managed for timber and has never been mined.

We established survey sites at open-canopy wetlands. We had four survey sites per land class, yielding a total of 16 survey sites. The sites ranged in elevation from 1100–1300 m. Except for three of the four survey locations for MAT, all survey points were inside the Mower Tract.

2.2.3. Bat Acoustic Surveys

We passively monitored bat echolocation calls using Song Meter SM3BAT detectors with external SMM-U2 ultrasonic microphones (Wildlife Acoustics, Maynard, MA). We separated the detector locations by at least 150 m to ensure each vocalization would only be recorded by one device (Agranat 2014). To minimize reflected echolocation calls from the water surface, we mounted microphones 3 m from the edge of the pools of water and 2 m above the ground on stakes. We angled microphones horizontally across the wetland and programmed the detectors to record ultrasonic signals above 16 kHz from sunset to sunrise. Detectors recorded for at least 3 seconds to a maximum of 15 seconds depending on a signal's duration. Recording occurred on four nights within June, July, and August of 2021 for a total of 12 nights of recordings per survey site. Because low temperatures, high winds, and rain can decrease insect and bat foraging activity (Burles et al. 2009), recordings only occurred on nights with temperatures above 10°C, wind speeds averaging below 8 km/h, and no significant precipitation. We used eight detectors and rotated them between the sixteen survey sites, randomly assigning detectors and microphones to their location during each sampling period.

We defined bat activity with three metrics: number of recordings, pulse counts, and feeding buzzes. We examined the acoustic recordings using Kaleidoscope Pro 5 (Wildlife Acoustics, Maynard, MA). We appraised all files for bat vocalizations. A permissible file required a minimum of one ultrasonic pulse. We removed files without any bat echolocation pulses, as they represented captures of ultrasonic waves from non-bat sources. Because individual recordings could have between 1 and >100 pulses, each recording varied in the amount of time and energy expended by the vocalizing bat. To compensate, we used Kaleidoscope to count individual echolocation pulses automatically in each recording. Finally, we manually counted feeding buzzes to evaluate foraging activity. Feeding buzzes are foraging attempts identified as a series of pulses emitted in quick succession (Russo et al. 2018).

Recordings with ≥ 5 pulses were identified to species using the software's bat call reference library and specifying Bats of North America 5.4.0 and West Virginia as the region. We excluded *Nycticeius humeralis* (evening bat) from the list because it is not known to occur at higher elevations in the mid-Atlantic (M. Ford, Virginia Tech, pers. comm.). We used the "+1 More Accurate" sensitivity setting, which produces fewer, but more accurate, identifications. We accepted identifications with presence p -value < 0.05 . Calls were labeled as "unidentified" if they did not meet these criteria or were of insufficient quality to be identified. Because *Myotis* species are difficult to differentiate using this program, we grouped all *Myotis* species together.

2.2.4. Insect Sampling

We conducted insect sampling via blacklight funnel traps (BioQuip Products, Rancho Dominguez, CA) at all 16 survey sites for one night per month during the study. Due to the number of traps available, insect sampling occurred over four to six nights each month. Insect sampling did not occur on the same nights the detectors were operating to avoid skewing the data. Trapping followed the same weather restrictions as acoustic surveying. We placed traps as close as possible to the recorder's designated location. Traps contained a dichlorvos-based insecticide strip and operated from sunset to sunrise via a timer. After sunrise, we collected and froze the samples. For each sample, we partitioned the insects into Lepidoptera and non-Lepidoptera

groups as Lepidopterans were considered primary prey items for bats in our study (Lacki et al. 2007, Dodd et al. 2015). Each group was counted, dried in an oven at 55° C for five days, and weighed to the nearest 0.1 mg on a Mettler Toledo AB204-S analytical balance (Mettler Toledo, Columbus, OH) to determine dry biomass.

2.2.5. Habitat Assessment

We recorded vegetation and aquatic characteristics of each wetland site to assess clutter, wetland size, and water quality on their potential covariance with bat and insect activity. To calculate pool surface area, pools were approximated as a circle, oval, or rectangle. We measured the diameter, the major and minor diameters, or the length and width, respectively, with a meter tape. We visually estimated percent cover of the pool surface area by woody debris, vegetation, and rocks. To calculate the volume of woody debris, we measured the length and diameters of the trunks and all the stems of the woody debris with a meter tape and calipers. A Levelogger 5 (Solonist, Georgetown, Canada) pressure transducer measured the hydroperiod of each wetland throughout the summer. Each Levelogger 5 was placed inside a slotted PVC pipe at the deepest spot of the pool and continuously logged water depth at 1-hour intervals.

We took a 250 mL water sample from each wetland in early summer, mid-summer, and early fall. Samples were frozen and transported to and analyzed at the UK Department of Forestry Hydrology Laboratory. Water quality testing included specific conductivity (SC $\mu\text{S}/\text{cm}$), sulfate (SO_4 mg/L), magnesium (Mg mg/L), calcium (Ca mg/L), potassium (K mg/L), sodium (Na mg/L), turbidity (NTU), pH (H+), nitrate (NO_3 mg/L), manganese (Mn mg/L), and iron (Fe mg/L). Lab pH was measured with an Orion Benchtop pH meter (Thermo Fisher Scientific, Waltham, MA). SC was measured using a YSI conductivity bridge (YSI, Yellow Springs, OH). Total Fe, Mn, Ca, K, Mg, and Na were measured using a GBC SDS 270 Atomic Adsorption Spectrophotometer (GBC Scientific Equipment, Melbourne, Australia). Nitrate was analyzed with a Brun Luebbe (Brun+Luebbe Company, Norderstedt, Germany) auto analyzer. Sulfate was measured using ion chromatography on a Dionex Ion Chromatograph 2000 (Dionex Corp., Sunnyvale, CA). Turbidity was measured with a Hach turbidimeter (Hach, Loveland, CO). All sampling, preservation, and analytic protocols followed those outlined in Greenberg et al. (1992).

Terrestrial habitat characteristics were centered at the bat detector's location. Within each quadrant of a 10 m radius plot, we measured the tallest vegetation with a meter tape, clinometer, or telescoping height pole. We considered a 10 m radius representative of the surrounding area as well as the distance of confident bat detection. We measured the distance to nearest road and contiguous forest edge as well as relative forest cover in a 40 m radius using 2013 and 2020 imagery on Google Earth Pro (v.7.3.4.8642, Google LLC, Mountain View, CA).

2.2.6. Statistical Methods

We used R 4.2.1 (R Core Team 2021, Vienna, Austria) for statistical analyses. The response variables for this study were the number of recordings, ultrasonic pulses, and feeding buzzes per night at each wetland. Because these response variables were overdispersed count data, we used a generalized linear mixed effects model with a negative binomial distribution for analyses. Modeling was done with the glmmTMB package (Brooks et al. 2017). Since sites were sampled across four consecutive nights, sample nights were temporally autocorrelated. To account for autocorrelation, we applied a first-order autoregressive correlation structure based on day number to the covariance for each wetland site, as site was the unit of replication in this experimental design. We undertook a two-stage analysis to understand the effects of habitat characteristics on bat activity. First, to evaluate the overall effect of land class on bat activity, we fit mixed models with a fixed effect of land class for all response variables. We used estimated marginal means to create pairwise comparisons of the land classes using the emmeans package (Lenth 2022). Second, to determine if local habitat characteristics and insect abundance influenced bat activity, four concept models were developed *a priori* and fit for each response variable. In selecting variables, predictors were tested for correlation using Spearman's rank test. Variables deemed strongly correlated ($r_s > 0.7$) were considered for exclusion (Schober et al. 2018). Average vegetation height, distance to nearest forest edge, and proportion of forest cover were highly correlated with each other. Of these variables, we retained distance to forest as an indicator of edge habitat, which is an area known to be used by bats. Day of year was included in all models to account for seasonality of activity relative to survey occurrence. Variables were scaled and centered prior to model fit to aid with model convergence and parameter comparison. The final models were LAND (distance to nearest forest edge and

distance to nearest road), WATER (surface area of the wetland pool, percent cover of surface area, volume of woody debris, and hydroperiod), INSECT (biomass of Lepidopterans – LEPID MASS, number of Lepidopterans – LEPID NUM, biomass of all insects – INSECT MASS, or number of all insects – INSECT NUM; analyzed independently), LAND + WATER, and GLOBAL (all variables included in the other three models). The models were evaluated using the Akaike Information Criterion (AIC) for small samples (AIC_c), difference in AIC_c from the top-ranking model (Δ AIC_c), and weights of models (AIC_w). Functions in the AICcmodavg package (Mazerolle 2020) were used to calculate these metrics. We considered models with Δ AIC_c differences ≤ 6 as minimally supported and Δ AIC_c ≤ 2 as good as the best supported model (Mazerolle 2006, Symonds and Moussalli 2011).

We were also interested in how the insect community varied with land class. Differences in biomasses and counts of total insects and Lepidopterans were assessed among the land classes. Negative binomial models were used for total insect and Lepidopteran counts, while total insect and Lepidopteran biomasses were modeled using the Gaussian distribution. Model fitting, selection, and inferential frameworks for these response variables were identical to those of bat activity.

Water quality parameters were statistically analyzed in R using the Kruskal-Wallis test with Dunn's test as a *post hoc* pairwise comparison between land classes when significance was found. All statistical tests were performed with a significance level of 0.05.

2.3. Results

2.3.1. Bat Acoustic Surveys

Each site recorded for 12 full nights producing 12,110 recordings that contained bat calls. Table 1 summarizes the total and average nightly production of the three activity indexes. Recordings and pulses per night were significantly greater in REGEN than FRA8 (recordings: $Z = -3.18$, $p = 0.008$, Cohen's $d = -0.47$; pulses: $Z = -3.25$, $p = 0.006$, Cohen's $d = -1.11$) (Figure 2.2 and Figure 2.3). FRA1 also had a greater number of recordings than FRA8, though this was marginally significant ($Z = 2.48$, $p = 0.063$, Cohen's $d = 0.37$). The recordings contained 1,800 feeding buzzes. With 61% of the feeding buzzes, REGEN had significantly greater foraging activity than the other three land classes (FRA1: $Z = -2.99$, $p = 0.015$, Cohen's $d = -0.75$; FRA8: $Z = -4.25$, $p < 0.001$, Cohen's $d = -1.06$; MAT: $Z = -2.88$, $p = 0.021$, Cohen's $d = -0.71$) (Figure 2.4). Of the 12,110 recordings, 6,282 sequences (51%) were identified to species. The following species were identified throughout the study area in decreasing order of occurrence: *Lasiurus borealis* (red bat; 45.3%), *Eptesicus fuscus* (big brown bat; 26.5%), *Lasiurus cinereus* (hoary bat; 21.2%), *Perimyotis subflavus* (tri-colored bat; 4.1%), *Lasionycteris noctivagans* (silver-haired bat; 2.2%), and *Myotis* spp. (0.7%) (Table 2.1). The *Myotis* species that can occur within our study area include *M. leibii* (small-footed myotis), *M. lucifugus* (little brown bat), *M. septentrionalis*, and *M. sodalis*. The identified *Myotis* sequences could be one or any combination of these four species.

2.3.2. Insect Survey

The black light traps captured 43,628 total insects weighing 232.9 g. Of these, 20,823 individuals weighing 178.4 g were identified as Lepidopterans. There were no significant differences among the land classes in the number of Lepidopteran individuals ($\chi^2 = 1.09$, $p = 0.78$), the Lepidopteran dry biomass ($\chi^2 = 0.22$, $p = 0.98$), the number of total insect individuals ($\chi^2 = 1.36$, $p = 0.71$), or the total insect dry biomass ($\chi^2 = 1.69$, $p = 0.64$) (Figure 2.5).

Table 2.1. Summary of bat activity. The nightly mean (n = 48 recording nights per land class) and standard deviation of recordings, pulses, and feeding buzzes for each land class.

	FRA1	FRA8	REGEN	MAT
Recordings				
Mean	92.5	28.4	86.9	44.5
SD	107.0	27.8	72.8	37.4
Total	4,439.0	1,363.0	4,170.0	2,137.0
Pulse Counts				
Mean	2,713.9	693.1	3,196.8	972.3
SD	3,989.8	812.8	3,027.2	811.7
Total	130,268.0	33,268.0	153,444.0	46,671.0
Feeding Buzz Counts				
Mean	7.7	2.3	22.9	4.7
SD	13.6	2.9	32.3	5.6
Total	367.0	112.0	1,098.0	223.0

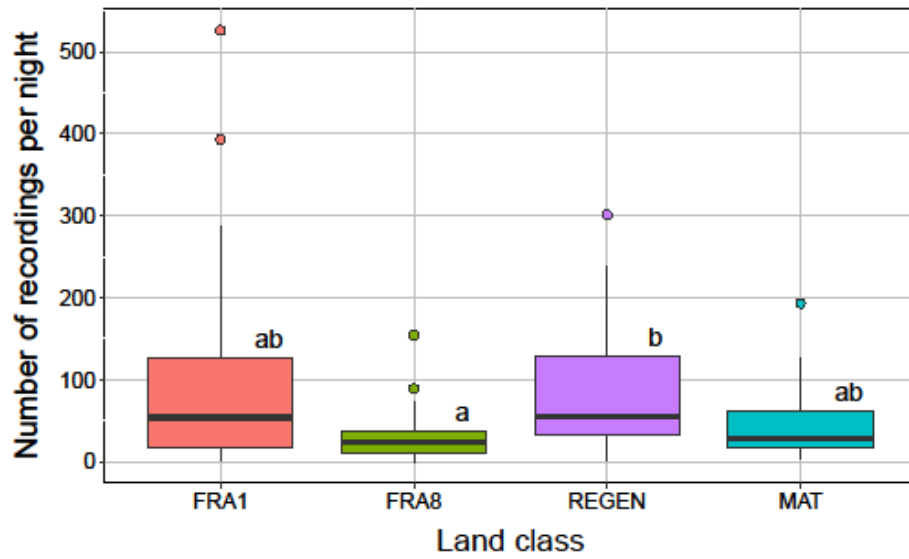


Figure 2.2. Boxplots showing the distribution of nightly recordings of each land class. Letters denote significance difference ($p < 0.05$).

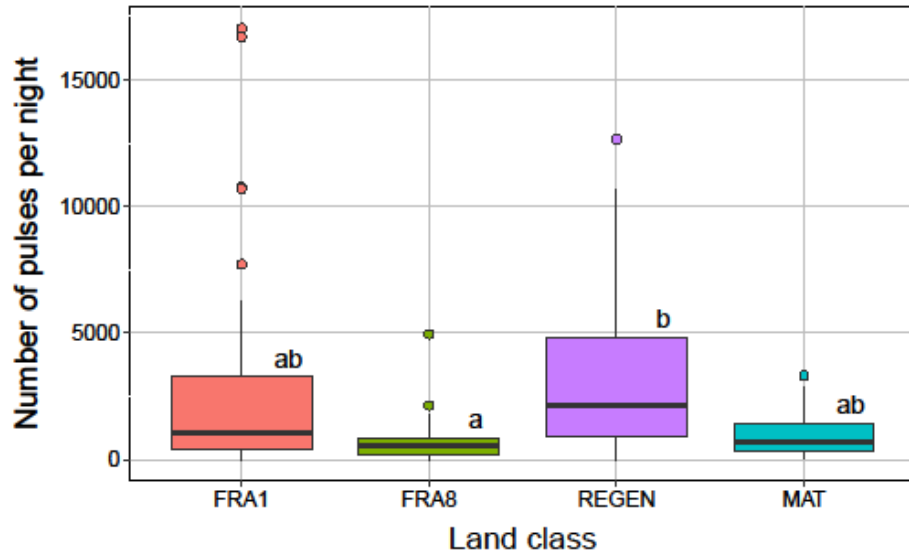


Figure 2.3. Boxplots showing the distribution of nightly pulse counts in each land class. Letters denote significance difference ($p < 0.05$).

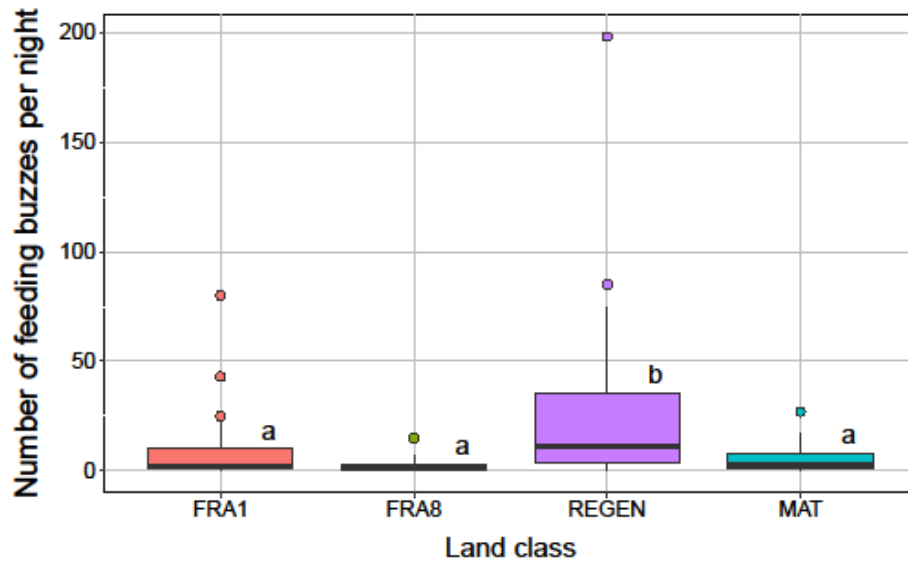


Figure 2. 4. Boxplots showing the distribution of nightly feeding buzzes in each land class. Letters denote significance difference ($p < 0.05$).

Table 2.2. Number of identified call sequences (≥ 5 pulses) for each bat species listed by land class. *Myotis* spp. potentially include *M. leibii*, *M. lucifugus*, *M. septentrionalis*, and/or *M. sodalis*.

Species	FRA1	FRA8	REGEN	MAT	Total
<i>Eptesicus fuscus</i>	1,388	37	153	91	1,669
<i>Lasiurus borealis</i>	820	318	1,184	522	2,844
<i>Lasiurus cinereus</i>	311	232	600	187	1,330
<i>Lasionycteris noctivagans</i>	0	66	0	73	139
<i>Myotis</i> spp.	5	5	34	0	44
<i>Perimyotis subflavus</i>	0	65	146	45	256
Total	2,524	723	2,117	918	6,282

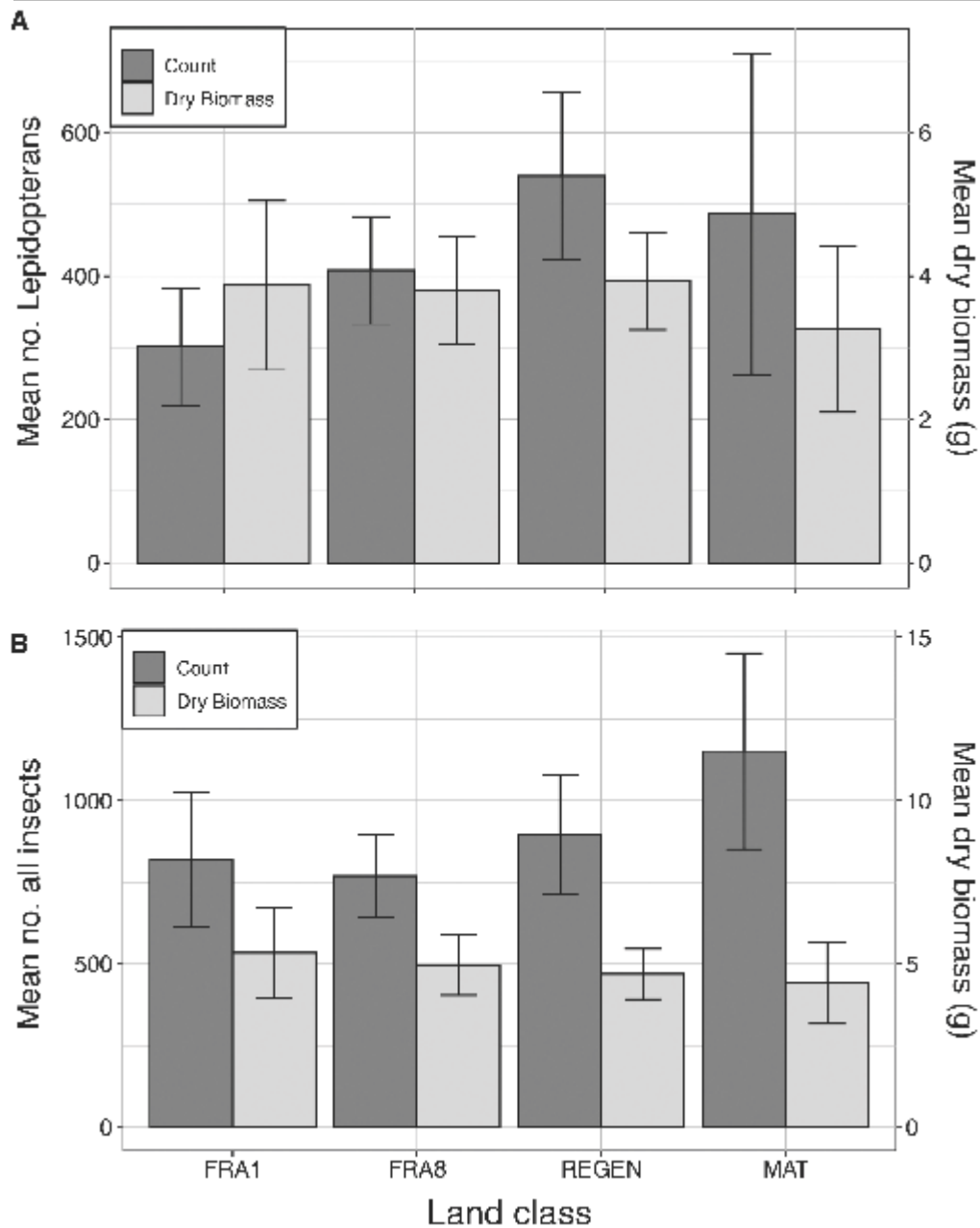


Figure 2.5. Comparison of Lepidopteran and total insect amounts by land class. A) Mean abundance of Lepidopterans sampled from each land class in counts of individuals and dry biomass (g) including standard error bars. B) Mean abundance of all insects sampled from each land class in counts of individuals and dry biomass (g) including standard error bars. There was not a significant difference in any category.

2.3.4. Habitat Assessment

The pool and vegetation characteristics are summarized in Table 3. FRA8 and REGEN pool surface areas varied by less than 1 m² on average. FRA1 pools were slightly larger, and MAT, with 66.7 m², had the largest average pool surface area. FRA8 wetlands had 55–75% of their pool surface areas covered and the highest average of 64% coverage. FRA1 and REGEN had less than half that percentage of cover on average and MAT had slightly more than half. REGEN had wetlands that were 4–13 m away from a contiguous forest edge with an average of 9.3 m, and MAT's wetlands were the second closest with a range of 2–56 m and an average of 18.3 m. FRA1 and FRA8 were over 50 m and 100 m on average, respectively, from a contiguous

forest edge. REGEN wetlands were the farthest from roads, an average distance of 428.5 m. FRA1 and MAT had large ranges that averaged 251.3 m and 223.8 m, respectively. At 144.5 m, FRA8 wetlands had the shortest average distance to the nearest road.

The water quality results are summarized in Table 4. Mg, Ca, K, pH, and Mn had significant variation (Kruskal-Wallis test; Mg: $H = 22.408$, $p < 0.001$; Ca: $H = 11.390$, $p = 0.01$; K: $H = 24.961$, $p < 0.0001$; pH: $H = 29.505$, $p < 0.0001$; Mn: $H = 25.587$, $p < 0.001$) among the land classes, though these parameters did not vary significantly between FRA1 and FRA8, nor did they vary significantly between REGEN and MAT. Rather, many parameters differed significantly between these two pairs of land classes. FRA1 and FRA8 had greater average levels for all parameters compared to both REGEN and MAT for specific conductance, Mg, Ca, K, turbidity, pH, and Mn. Fe, Mn, SO_4 , and conductivity are parameters of particular concern on mined lands. Mn ranged from 0–10 mg/L with means from 0.43–1.89 mg/L for the three previously mined land classes. MAT had lower levels with a Mn range of 0–0.13 mg/L and a mean of 0.07 mg/L. Fe ranged from 0–21 mg/L with means from 0.46–2.47 mg/L for all land classes. SO_4 ranged from 1–29 mg/L with means from 7.31–16.01 mg/L for all land classes. Conductivity ranged from 10–153 mg/L with means from 21.03–53.87 mg/L for all land classes.

None of the wetland pools ran dry during the study, so hydroperiod was excluded from the WATER model parameters. FRA8 had the shallowest pools on average, with depths between 24.8–54.2 cm. Like FRA8, FRA1 had depths which averaged between 30.4–54 cm. REGEN had the second greatest pool depths, which averaged between 41.2–59.1 cm. MAT, with average depths between 51.9–73.9 cm, had the deepest pools on average.

LAND was the top model of the candidate set for all three bat activity types (Table 2.3), although there was one exception for the number of pulses in which LEPID NUM was the best model ($AIC_w = 0.49$) and LAND was the second-best model, though equally supported ($\Delta AIC_c = 0.16$, $AIC_w = 0.45$). In most cases, LAND had an AIC_w of at least 0.5 more than the next best model. Within LAND, neither distance to forest nor distance to road were significant in explaining the variation in number of recordings or number of pulses, but feeding buzzes significantly increased as distance to nearest forest decreased ($Z = -2.22$, $p = 0.027$). For the number of recordings, LEPID NUM and INSECT MASS also had strong support, and WATER, LEPID MASS, and INSECT NUM had minor support (Table 2.3). From WATER, greater pool cover was associated with significantly fewer recordings ($Z = -2.32$, $p = 0.020$). LEPID MASS, INSECT MASS, INSECT NUM, and GLOBAL had minor support as models for the number of pulses (Table 2.3), although only the number of Lepidopterans and total biomass of insects were significant in explaining variation ($Z = -2.76$, $p = 0.006$; $Z = -2.12$, $p = 0.034$; respectively). LEPID NUM, INSECT MASS, and INSECT NUM had minor support as models for the number of feeding buzzes (Table 2.3) with only total biomass of insects significantly explaining variation ($Z = -2.04$, $p = 0.041$). All insect variables decreased as the response variables increased. All response variables increased with day of year ($Z = 2.12$ – 4.54 , $p = <0.001$ – 0.034).

Table 2.3. Model selection for the three response variables. Model selection was based on Akaike Information Criteria (AIC) with correction for small sample sizes (AIC_c), difference in AIC (ΔAIC_c) between the current model and the top ranked model, and model weight (AIC_w). K is the number of parameters in each model. Models with a $\Delta AIC_c \leq 6$ have minimal support and $\Delta AIC_c \leq 2$ as good as the best supported model.

Model Sets		K	AIC_c	ΔAIC_c	AIC_w
Recording Counts	LAND	7	1893.72	0.00	0.71
	LEPID MASS	6	1896.57	2.85	0.17
	WATER	8	1897.48	3.76	0.11
	WATER + LAND	10	1902.17	8.45	0.01
	GLOBAL	11	1906.26	12.54	0.00
	LAND	7	1893.72	0.00	0.54
	LEPID NUM	6	1894.55	0.83	0.36
	WATER	8	1897.48	3.76	0.08
	WATER + LAND	10	1902.17	8.45	0.01
	GLOBAL	11	1903.61	9.89	0.00
	LAND	7	1893.72	0.00	0.58
	INSECT MASS	6	1894.86	1.14	0.33
	WATER	8	1897.48	3.76	0.09
	WATER + LAND	10	1902.17	8.45	0.01
	GLOBAL	11	1904.78	11.06	0.00
LAND	7	1893.72	0.00	0.67	
INSECT NUM	6	1896.02	2.30	0.21	
WATER	8	1897.48	3.76	0.10	
WATER + LAND	10	1902.17	8.45	0.01	
GLOBAL	11	1905.27	11.55	0.00	
Pulse Counts	LAND	7	3163.07	0.00	0.85
	LEPID MASS	6	3167.46	4.39	0.09
	WATER	8	3169.57	6.50	0.03
	WATER + LAND	10	3171.16	8.09	0.01
	GLOBAL	11	3173.48	10.41	0.00
	LEPID NUM	6	3162.91	0.00	0.49
	LAND	7	3163.07	0.16	0.45
	GLOBAL	11	3167.90	4.99	0.04
	WATER	8	3169.57	6.66	0.02
	WATER + LAND	10	3171.16	8.25	0.01
	LAND	7	3163.07	0.00	0.70
	INSECT MASS	6	3165.18	2.11	0.25
	WATER	8	3169.57	6.50	0.03

	WATER + LAND	10	3171.16	8.09	0.01
	GLOBAL	11	3171.49	8.42	0.01
	LAND	7	3163.07	0.00	0.81
	INSECT NUM	6	3166.58	3.51	0.14
	WATER	8	3169.57	6.50	0.03
	WATER + LAND	10	3171.16	8.09	0.01
	GLOBAL	11	3172.16	9.09	0.01
Feeding Buzz Counts					
	LAND	7	1107.98	0.00	0.94
	WATER + LAND	10	1115.39	7.41	0.02
	LEPID MASS	6	1115.53	7.54	0.02
	WATER	8	1116.63	8.65	0.01
	GLOBAL	11	1117.94	9.96	0.01
	LAND	7	1107.98	0.00	0.88
	LEPID NUM	6	1113.42	5.44	0.06
	GLOBAL	11	1115.03	7.04	0.03
	WATER + LAND	10	1115.39	7.41	0.02
	WATER	8	1116.63	8.65	0.01
	LAND	7	1107.98	0.00	0.84
	INSECT MASS	6	1112.23	4.25	0.10
	GLOBAL	11	1115.05	7.06	0.02
	WATER + LAND	10	1115.39	7.41	0.02
	WATER	8	1116.63	8.65	0.01
	LAND	7	1107.98	0.00	0.89
	INSECT NUM	6	1113.56	5.58	0.05
	GLOBAL	11	1115.16	7.17	0.02
	WATER + LAND	10	1115.39	7.41	0.02
	WATER	8	1116.63	8.65	0.01

2.4. Discussion

Bats are utilizing FRA-restored mined lands in both the 1-year and 8-year age classes. We recorded commuting activity as well as foraging activity. FRA1 had greater average activity than FRA8 for all three response variables. Both restored land classes had activity levels similar to those from MAT, though FRA1 had greater activity than MAT and FRA8 had lower activity than MAT. However, REGEN had significantly more foraging activity than the other land classes.

Insect and Lepidopteran abundances and biomass were statistically indistinguishable across the land classes, suggesting that FRA restoration practices did not hinder the establishment of a prey base for bats. Despite the similar quantities, modeling revealed they had a minor role in explaining bat activity levels as they were almost always the second-best model. The number of Lepidoptera and total insect biomass were significant parameters for pulse activity and total insect biomass was a significant parameter for feeding buzzes. However, the relationship between the insect variables and bat activity indexes was negative, indicating that bats were still active at our survey wetlands despite decreased insect abundances. Wolbert et al. (2014) also found a negative relationship between bat activity and insect abundance, which they attributed to

peak bat activity occurring at lower temperatures than peak insect biomass sampling. Our result most likely occurred for different reasons. In our study, there was a significant increase in bat activity throughout the summer across the three activity indexes. The increased activity could be associated with young bats becoming volant in July (Lacki et al. 2007). The increased bat activity, and thus increased prey consumption, could have caused smaller insect samples, but this is not an occurrence we could truly detect with our study design.

Another possible explanation is lower insect abundances could cause longer foraging bouts (Wilkinson and Barclay 1997), creating a greater number of recordings and foraging attempts.

The effect of insect abundance on bat activity is varied in the literature. In South Carolina, the diversity and abundance of insects did not affect bat activity, while vegetation and water salinity did prove influential (Moore and Best 2018). Grindal and Bringham (1999) measured similar insect availability between forest edges and forest interior and significantly lower availability in clearcuts, yet bats had a much greater foraging rate only at the edge habitat. In a study of ponds, meadows, and clearcuts, overall bat activity was greater at ponds, but foraging activity was correlated with higher insect abundance at meadows and clearcuts (Seibold et al. 2013). The degree of importance of insect abundance is unclear, but bats cannot forage in areas without prey. The ability of restored mined land to provide foraging habitat beginning at least one year post-restoration and as the site matures is thus of great consequence.

Bat activity was best explained by landscape attributes. Activity decreased with proximity to roads and significantly increased with proximity to contiguous forest edges. REGEN not only had the farthest average distance to the nearest road but also the shortest average distance to a forest edge. REGEN's relation to forest edges makes it unique among the land classes. REGEN's areas are narrow strips of land, on average 30 m wide. This land class then acts like a linear corridor, whereas MAT are wetlands within small openings in mature forest and the FRA-restored areas are large, open expanses. Bats use edges for navigating and commuting (Zimmerman and Glanz 2000, Law and Chidel 2002, Murray and Kurta 2004), foraging (Krusic et al. 1996, Grindal and Brigham 1999, Jantzen and Fenton 2013, Langridge et al. 2019), and roosting (Barclay and Kurta 2007, Law et al. 2016). REGEN thus combines the benefits of edge habitat with the prey availability of wetlands, which could explain its high activity levels. Site 4 from FRA1 had a similar physical layout to REGEN (Figure 2.1). Unlike the other three sites of FRA1, which were located in large, open expanses of restored area, site 4 was positioned in an 80 m wide strip of restored area bordered by mature forest. This corridor-like nature may have contributed to the disproportionate bat activity observed there: 56% of the recordings and 70% of the pulses and feeding buzzes from FRA1. Another possible explanation is a roost site may exist nearby, particularly of *E. fuscus* as they were the most numerous identified for that site.

Cover of the pool surface contributed to variation in bat recordings. FRA8 may have had less activity than the other land classes because of increased clutter on the water's surface. On average, FRA8's pools had 64% of their surface area covered by vegetation, rocks, and woody debris, whereas the other land classes had 26–39% cover on average. Moore and Best (2018) found that bats were significantly more active over open water wetlands than wetlands with vegetation cluttering the water. Bats prefer foraging in uncluttered habitats as clutter produces extraneous background noise when hunting for insects (Mackey and Barclay 1989). Cattails of approximately 2 m in height comprised most of the pool cover for FRA8. FRA1 wetlands, having only recently been created, did not have enough time to develop cattails within its pools, so they remained relatively open. REGEN, likewise, had open pools that lacked cattails. MAT's pools contained wetland grass species, but the grasses had a maximum height of 0.5 m.

During our study, we detected *L. borealis*, *L. cinereus*, *L. noctavigans*, *E. fuscus*, *P. subflavus*, and *Myotis* species. *L. borealis*, *L. cinereus* and *L. noctavigans* have not experienced mortality from WNS since they are long-distance migrants that do not hibernate in caves (Hoyt et al. 2021). These species were expected to have a greater number of detections than the other species. *L. borealis* and *L. cinereus*, which forage and travel in both open areas and edges (Loeb and O'Keefe 2011), appropriately had high detection levels across the study area. *E. fuscus* will travel and forage in forests, at edges, or in openings, so it was not surprising to detect them in each land class. *L. noctavigans*, however, was only detected in FRA8 and MAT and had 1/10th and 1/20th the number of detections as *L. borealis* and *L. cinereus*, respectively. *L. noctavigans* tends to travel and forage in open areas (Loeb and O'Keefe 2011), so determining why *L. noctavigans* were not detected in FRA1 and why they used MAT requires further study. *Myotis* spp. were mostly found in REGEN, though detection levels were low. Because most *Myotis* spp. prefer forested areas and edge habitat (Loeb and O'Keefe 2011), their occurrence could be explained by the corridor-like nature of the land class and the mature forest in

the surrounding area.

Overall, water quality at the wetlands in FRA1 and FRA8 was good and significantly better than reported for streams impacted by mountain top mining in the region (Muncy et al. 2014, Price et al. 2016). Conductivity, pH, Ca, Mg, and K were slightly higher at FRA sites, which could be due to the nature of some of the freshly plowed spoils, whose unweathered rock can leach ions when exposed to water (Agouridis et al. 2012). Mn, Fe and SO₄, which are all parameters of concern on coal mine sites, were low. While there is concern for bats drinking toxic water on mined lands (Korine et al. 2016, Frick et al. 2019), the water in created wetlands on FRA-restored sites have not exhibited toxicity. Additionally, in the Central Appalachians, a specific conductivity threshold of 300 µS/cm is deemed protective of aquatic biota (US EPA 2011), and all of the examined treatment means were well below that level. The water quality and hydroperiod results from the FRA wetlands indicate they may be useful to wildlife and not cause detrimental impacts. Lambert et al. (2021) credited good water quality as a factor for amphibian occupancy in created wetlands at the Mower site.

2.4.1. Management Implications

Reforestation of reclaimed legacy mines that are in a state of arrested succession will certainly help restore lost ecosystem function, but reforestation complemented with wetland creation will provide further ecosystem benefits. The created wetlands in this study area exhibited good water quality, increased plant biodiversity (Branduzzi et al. 2020), and provided suitable habitat for amphibians (Lambert et al. 2021). Our results indicate that bats are also using the restored mined lands as habitat, including utilizing created wetlands as foraging locations. The edge habitat provided by narrow corridors of restored land could be especially beneficial for bats, particularly if these corridors contain wetlands. Created wetlands can be susceptible to invasion by non-native and/or undesirable plant species, however, necessitating vegetation management to maintain the wetlands' suitability for bat foraging. As the restored lands age, the edge habitat will give way to mature forest. The created wetlands, however, add to the mosaic of the forested landscape and can continue to offer bat foraging habitat.

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Section 3. Response of Pond-breeding Amphibians to Wetland Creation and Reforestation of Legacy Surface Mines in the Central Appalachian Region

3.1. Introduction

Amphibian populations have experienced significant declines in recent decades. As a result, amphibians are considered to be the most threatened class of vertebrates (Luedtke et al. 2023). Most population declines are attributed to habitat loss and degradation exacerbated by the recent effects of climate change (Luedtke et al. 2023; Sodhi et al. 2008). For pond-breeding amphibians, high rates of seasonally inundated wetland loss (Calhoun et al. 2017) and climate-driven changes to wetland hydroperiod pose significant limitations to reproductive success and population viability (Walls et al. 2013). The creation of new wetlands could benefit pond-breeding amphibians by providing additional breeding sites across a landscape (Shoo et al. 2011; Miller et al. 2018). Furthermore, terrestrial habitat adjacent to wetlands is critical for the persistence of many pond-breeding species (Semlitsch et al. 2003). These terrestrial habitats provide microhabitats and microclimates suitable for activities during the non-breeding season such as foraging, overwintering, dispersing, and migrating (Semlitsch et al. 2000). As a result, the restoration of terrestrial habitat adjacent to created wetlands may be an important climate adaptation strategy for pond-breeding amphibians.

High elevation red spruce (*Picea rubens*) forest ecosystems in the Central Appalachians are inhabited by at least nine pond-breeding amphibian species (Lambert et al. 2021), yet these forests are at risk due to resource extraction and climate change. Red spruce-dominated forests experienced range reductions from timber harvest in the late 1800s, followed by coal surface mining during the late 1900s (Byers et al. 2010). In West Virginia (USA) alone, red spruce-dominated forests currently occupy approximately 10% of their historic range (Byers et al. 2010). Additional range reductions are expected due to climate change, with an 85% reduction in red spruce-suitable habitat predicted for West Virginia by the year 2080 (Beane et al. 2015). In addition to the direct impacts on forest composition and structure from climate change, changes in temperature and precipitation regimes will likely impact high elevation wetlands, given that their hydrology is highly dependent on snow melt and precipitation (Burkett et al. 2000). Consequently, the negative synergistic effects of habitat loss and climate change in these sensitive ecological areas is likely to decrease amphibian population size and distribution.

In response to the large-scale reduction of red spruce forest ecosystems in the Appalachians, land managers have implemented intensive forest restoration projects in degraded areas including reclamation efforts on legacy surface coal mines in the Central Appalachian region (Rhodes 2022). Specifically, the Forestry Reclamation Approach (FRA) has been used to restore red spruce-dominated forests on legacy minelands in the Monongahela National Forest (MNF), West Virginia. The FRA utilizes non-native vegetation removal, soil decompaction, woody debris loading, planting of native trees and shrubs, and creation of wetlands to restore ecosystem function to native forests (Burger et al. 2013; Branduzzi et al. 2022; Groninger et al. 2007).

By using best industry practices in forestry and restoration ecology, the FRA encourages the succession of native forests at faster rates compared to natural succession on sites reclaimed as grasslands (Burger et al. 2013). The FRA could also enhance habitat for pond-breeding amphibians by increasing breeding sites and microclimate and microhabitat refugia adjacent to breeding sites; however, amphibian communities and abundances on FRA sites and those in undisturbed forests have yet to be compared.

Herein, we assessed pond-breeding amphibian response to wetland creation and reforestation (via the FRA) on legacy surface mines in the MNF, West Virginia. Our first objective was to compare wetland attributes (e.g., size, canopy cover, water chemistry) across four forest treatments including younger FRA sites (i.e., those reclaimed 2–5 years ago); older FRA sites (i.e., those reclaimed 8–11 years ago); sites initially reclaimed as grasslands that were left to undergo natural forest succession; and unmined mature forests. Our second objective was to compare pond-breeding amphibian occupancy and species richness in wetlands across the four treatments. Finally, our third objective was to compare abundances of four commonly occurring amphibian species (Spotted Salamander (*Ambystoma maculatum*), Wood Frog (*Lithobates sylvaticus*), Green Frog (*Lithobates clamitans*), and Eastern Newt (*Notophthalmus viridescens*)) across the four treatment types. By examining wetland attributes, amphibian species occupancy, amphibian species richness, and species abundance across legacy surface mines, we aimed to assess the capacity of wetland creation and forest reclamation to improve habitat availability for pond-breeding amphibians.

3.2. Materials and Methods

3.2.1. Study Sites

Study sites were located on legacy minelands at the Mower Tract (1478 m elevation; Randolph County) and Sharp Knob (1382 m elevation; Pocahontas County) in the MNF (See Section 1, Figure 1.1). Historically, our study area was characterized as red spruce–northern hardwood forest, with poor soils and a thick peat layer, and scattered, isolated wetlands (Byers et al. 2010). The region is often immersed in cloud cover with a mean precipitation of 4140 mm/yr (Byers et al. 2010). Our study sites experienced significant logging from the 1880s to the 1920s, followed by extensive surface coal mining in the 1970s and 1980s (Byers et al. 2010). After mining, the land was recontoured and reclaimed as grassland via the planting of non-native species (i.e., tall fescue (*Festuca arundinacea*), sericea lespedeza (*Lespedeza cuneata*)). Native forests struggled to reestablish, leaving the landscape in a state of arrested succession (Groninger et al. 2007). Following grassland reclamation, the land was sold to the US Forest Service and incorporated as part of the MNF. Beginning in 2010, the US Forest Service partnered with Green Forests Work and the Appalachian Regional Reforestation Initiative to restore red spruce–northern hardwood forests using FRA on these legacy minelands. As part of the restoration plans, non-native species were removed, soils were decompacted using a deep ripping shank, coarse woody debris was loaded into the project areas, and over 800,000 native trees and shrubs were planted. In addition, approximately 800 shallow (<0.5 m), seasonal wetlands were created. See Lambert et al. (2021) for more information on the project area.

3.2.2. Treatment Types

To examine amphibian response to restoration and wetland creation, we identified 32 wetlands located on or near legacy surface mines at the Mower Tract (n = 29) and Sharp Knob (n = 3) (See Section 1, Figure 1.1). We chose wetlands that often dry during late summer or fall (i.e., seasonally inundated wetlands), as these wetlands are generally devoid of many amphibian predators, particularly fish, and are considered a preferred breeding habitat for most pond-breeding amphibian species.

We selected wetlands within four treatment categories: younger FRA (YFRA) sites (2–5 years post-reclamation), older FRA (OFRA) sites (8–11 years post-reclamation), naturally regenerated (REGEN) sites (>40 years since mining) that were originally reclaimed as grasslands that were left to undergo natural forest succession, and unmined, mature (MAT) sites (Figure 3.1). YFRA sites were replanted with a mix of red spruce and native hardwoods between 2017 and 2020. Planted trees averaged < 1 m in height during data collection in 2022. Herbaceous species, including swamp milkweed (*Asclepias incarnata*) and boneset (*Eupatorium perfoliatum*), were planted around the constructed wetlands. OFRA sites were replanted with red spruce, aspen (*Populus* spp.), serviceberry (*Amelanchier arborea*), and black cherry (*Prunus serotina*) from 2011–2014. At the time of surveying in 2022, most of these planted seedlings were between 2–4 m high. We considered both YFRA and OFRA to be young successional forest as herbaceous plant cover was extensive both around the wetlands and within the surrounding uplands. All wetlands in FRA treatments were constructed during the site preparation (i.e., soil decompaction) and were created using a mid-sized excavator to create a depression in areas with clay or wet soils. Small berms surrounded created wetlands to promote inundation, and downed trees, other woody debris, and large rocks were placed in wetlands as habitat features. Woody debris was loaded in the terrestrial areas around wetlands. Naturally regenerated sites (REGEN) were

mined prior to the Surface Mining Control and Reclamation Act of 1977 (SMCRA) and initially reclaimed as grasslands. These sites were located on coal mine benches between steep highwalls on flat, narrow strips of land with native forest adjacent to mined areas. The compacted soils at REGEN sites limited the growth of native trees, resulting in extensive herbaceous ground cover [19]. Non-native conifers (e.g., Norway spruce (*Picea abies*)) were planted at a few of the sites after initial grassland reclamation, resulting in partial or complete canopy closure over wetlands. Finally, MAT sites were wetlands within mature, second growth red spruce-northern hardwood forest that were not impacted by coal surface mining. Red spruce, yellow birch (*Betula alleghaniensis*), red maple (*Acer rubrum*), and beech (*Fagus grandifolia*) were the dominant tree species at the MAT sites.

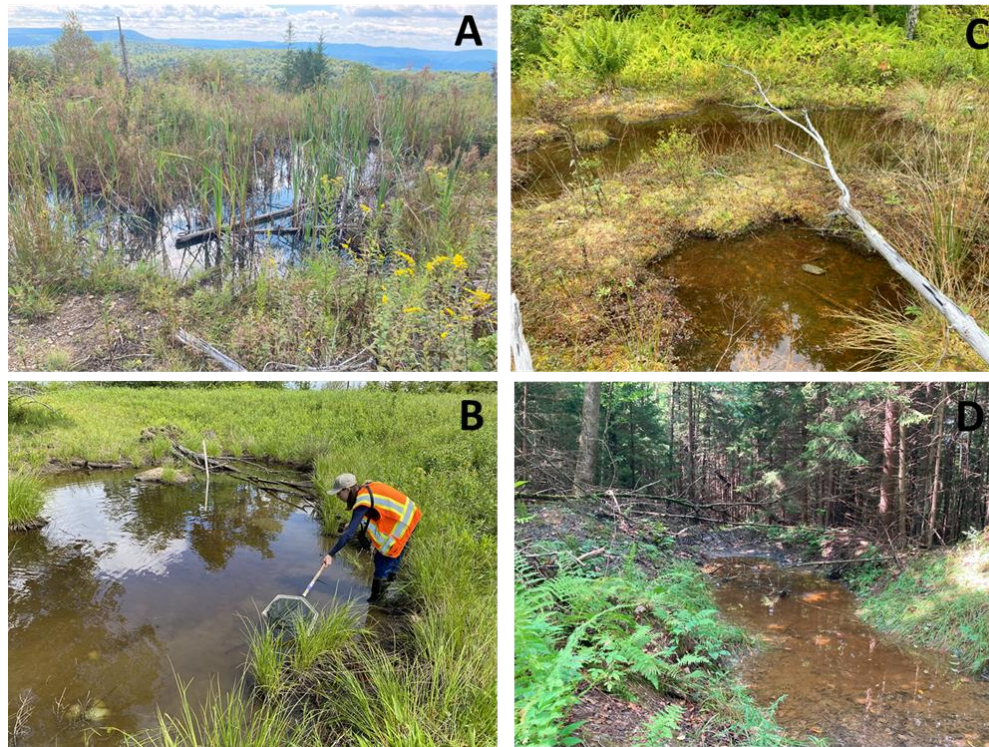


Figure 3.1. Examples of treatment types and wetlands in the Monongahela National Forest (West Virginia, USA). Wetland attributes and amphibian data were collected at (A) 2–5 year old sites restored via the Forestry Reclamation Approach; (B) 8–11 year old sites restored via the Forestry Reclamation Approach; (C) wetlands in naturally regenerated (REGEN) minelands (>40 years old); and (D) wetlands in unmined, mature (MAT) forests.

3.2.3. Wetland Attributes

Prior to amphibian sampling, we collected several physical measurements at each wetland to examine how wetland dimensions, water chemistry, and canopy cover varied across treatment types. We measured wetland length and width at the widest points of the basin, and surface area (length \times width). Canopy cover was estimated in the middle of June 2022 using a spherical crown densiometer (Forestry Suppliers, Jackson, MS, USA), calculated as the average of four measurements taken facing each cardinal direction while standing in the approximate center of the wetland. We collected a 100 mL water sample from each wetland on three occasions (~once per week) during the field season. Water samples were kept on ice in the field, and frozen until they were transported to the University of Kentucky Department of Forestry and Natural Resources Hydrology lab. All water samples were analyzed for turbidity (FTU), conductivity ($\mu\text{S cm}^{-1}$), pH (H^+), Total Organic Carbon (TOC (mg/L)), Cl (mg/L), SO_4 (mg/L), $\text{NO}_3\text{-N}$ (mg/L), $\text{NH}_4\text{-N}$ (mg/L), Ca^{+2} (mg/L), Mg^{+2} (mg/L), K^+ (mg/L), Na^+ (mg/L), Mn (mg/L), Fe (mg/L), Al (mg/L), and $\text{NO}_2\text{-N}$ (mg/L). Water quality sampling, preservation, and analytic protocols were performed in accordance with standard methods (Greenberg et al. 1992).

3.2.4. Amphibian Surveys

We conducted dip net surveys for amphibians from 8 to 30 June 2022. Sweeps were performed according to the protocol of Denton and Richter (2013), in which each sweep encompassed approximately 5 m of the wetland perimeter. To standardize the procedure, each wetland was measured to calculate the perimeter, which was then divided by 5. A 40 × 23 cm D-frame dip net was dragged across the bottom of the wetland for about one meter per sweep. The contents of the net were transferred into sorting bins, and all amphibian specimens were counted, identified to species, and returned to the wetland at the approximate location of capture. We could identify most individuals to species using adult or larval characteristics; however, members of the gray treefrog complex (i.e., Gray Treefrog (*Hyla versicolor*) and Cope’s Gray treefrog (*H. chrysoscelis*)) are identical in appearance and must be identified via genetics, cytology, or trill rate. Nocturnal audio recordings (unpublished data) confirmed that *H. versicolor* is the only member of the gray treefrog complex at our study sites. Each site was surveyed four times throughout the season (~once a week).

3.2.5. Data Analysis

3.2.5.1. Wetland Attributes

We performed an analysis of variance (ANOVA) in R (version 4.2.2; (R Core Team 2024) to examine differences among wetland attributes (i.e., wetland area, canopy cover, water chemistry) among our 4 treatment types (i.e., YFRA, OFRA, REGEN, MAT). Significance was assessed using an alpha level of 0.05. We performed a Tukey’s Honestly Significant Difference (HSD) test to further analyze significant differences detected by the ANOVA.

3.2.5.2. Multi-Species Occupancy Models

To examine the effect of treatment type and wetland area on amphibian occupancy and species richness, we used a Bayesian multi-species occupancy model (MacKenzie et al. 2017; Zipkin et al. 2009). This model generates mean occupancy across all species, occupancy estimates (Ψ) and detection probability (p) on a species-specific level, and species richness estimates (SpR) for each individual site and treatment type. The multi-species occupancy modeling approach combines single species estimates with the average parameter estimate for the entire community. This approach reduces bias and improves the precision of parameter estimates for species with few detections by borrowing information from species with more detections (Pacifiçi et al. 2014). However, borrowing information is only appropriate for species that share ecological, functional, or behavioral relatedness (Pacifiçi et al. 2014). We included all detected species in the same model because of their shared preference for using seasonally inundated wetlands as breeding habitat.

Detection/non-detection data were formatted into a matrix (i, j, k), where species i was detected at site j on sampling occasion k . Occupancy probability (Ψ_{ij}) represented the probability of species i occurring at site j , modeled as a function of covariate parameters on α . Detection probability, p_{ijk} , represented the probability of detecting species i at site j on sampling occasion k , modeled as a function of covariate parameters on β . We modelled multi-species occupancy with the following equation:

$$\text{logit}(\Psi_{ij}) = u_i + \alpha_{1i}(\text{YFRA})_j + \alpha_{2i}(\text{OFRA})_j + \alpha_{3i}(\text{REGEN})_j + \alpha_{4i}(\text{zArea})_j$$

Detection probability was modeled on the following equation:

$$\text{logit}(p_{ijk}) = v_i + \beta_{1i}(\text{Date})_j$$

Parameters α_{1i} , α_{2i} , α_{3i} , were the effect of treatment types (YFRA, OFRA, and REGEN, respectively), with “MAT” serving as the reference category. Parameter α_{4i} , was the effect of the continuous covariate area (i.e., wetland area), standardized to have a mean of zero. The parameter β_{1i} was the effect of date on amphibian detection probability. We excluded canopy cover from our model as canopy cover was highly correlated with treatment type (see Section 3.3.1 Wetland Attributes in Results). We estimated species richness (SpR) for each study site by summing indicator variables for occupancy for each amphibian species at each site for each model iteration and generating a posterior predictive distribution for species richness in each treatment.

We used Markov chain Monte Carlo sampling within a Bayesian modelling framework (Link et al. 2002). Priors were uninformative and uniformly distributed with a minimum of -3 and a maximum of 3 (i.e., $U(-3, 3)$ for α , β , and community-level parameters), and $U(0, 5)$ for all standard deviation (σ) parameters.

Three parallel chains were run, and convergence was assessed with the Gelman–Rubin statistic; all models were below 1.02, indicating model convergence (Gelman and Rubin 1992). Each chain was run for 200,000 iterations with a burn-in of 20,000 samples thinning rate of 3 (i.e., retained every 3rd sample). Thus, model output resulted in 60,000 samples and we summarized the posterior distribution, from which we calculated the mean, standard deviation, and 95% credible intervals (CI). We considered parameter estimates with credible intervals that did not contain zero as biologically meaningful variables informing occupancy and detection probability. We executed this model in Program R Version 4.2.2 with package R2WinBUGS (Sturtz et al. 2005), which exports data into WinBUGS Version 1.4 (Spiegelhalter et al. 2003).

3.2.5.3. *N-Mixture (Abundance) Models*

We used N-mixture models (Royle 2004) to examine the effects of site-specific covariates (i.e., treatment type and wetland area) and the sampling covariate, date, on species-specific abundance. For this analysis, we estimated abundance of the four most commonly detected species, including Spotted Salamander, Wood Frog, Green Frog, and Eastern Newt. Using count data from our four replicate surveys (c_{ij}), we modeled counts as independent outcomes of binomial sampling with index N_i and detection probability p_j . Site-specific abundances (λ) were modeled with a Poisson distribution, and heterogeneity in abundance among populations due to site covariates (x_i) were modeled using a Poisson-regression formulation of local mean abundances as:

$$N_i | \lambda_i \sim \text{Poi}(\lambda_i)$$

$$\log(\lambda_i) = \beta_1 + \beta_2 (\text{YFRA})_i + \beta_3 (\text{OFRA})_i + \beta_4 (\text{REGEN})_i + \beta_5 (\text{zArea})_i$$

We examined heterogeneity in detection (p_i) due to sampling covariates (x_{ij}) as:

$$c_{ij} | N_i \sim \text{bin}(N_i, p_{ij})$$

$$\text{logit}(p_{ij}) = \alpha_1 + \alpha_2 (\text{Date})_j$$

Parameters β_1 , β_2 , β_3 , β_4 , and β_5 were effect of treatment types (YFRA, OFRA, and REGEN, and zArea, respectively), with “MAT” serving as the reference category. The parameter α_2 was the effect of date on amphibian detection probability.

Similar to the multi-species occupancy model, all models used uninformative priors. Specifically, β_1 , β_2 , β_3 , β_4 , and $\beta_5 \sim N(0, 10^2)$, α_1 and $\alpha_2 \sim N(0, 10^2)$. We used Markov chain Monte Carlo sampling, and three parallel chains were run for each species model; convergence was assessed with the Gelman–Rubin statistic to ensure all models were below 1.02 (Gelman and Rubin 1992). Each chain was run for 200,000 iterations with a burn-in of 100,000 samples, and thinning rate of 3. Thus, model output was 60,000 samples and we summarized the posterior distribution, from which we calculated the mean, standard deviation, and 95% CI. We executed this model in Program R Version 4.2.2 with package R2WinBUGS (Sturtz et al. 2005), which exports data into WinBUGS version 1.4 (Spiegelhalter et al. 2003).

3.3. Results

3.3.1. *Wetland Attributes*

Among the wetland attributes we measured, we found only three significant differences across treatment types (Table 1). Canopy cover percentages were different between the treatment types ($F_{3,28} = 5.67$, $p = 0.004$), with REGEN and MAT sites having significantly more canopy cover than the YFRA sites. The pH of the water was different among treatment types ($F_{3,28} = 6.71$, $p = 0.001$), with YFRA wetlands and OFRA wetlands having higher pH values than REGEN sites. We also found that TOC differed across treatment types ($F_{3,28} = 3.76$, $p = 0.02$), with OFRA sites having higher TOC than REGEN sites. Differences in aluminum concentrations between treatments were marginally significant ($F_{3,28} = 2.87$, $p = 0.06$). Wetland area did not differ between treatment types ($F_{3,28} = 0.44$, $p = 0.73$) and water chemistry variables that are often found to be elevated in surface waters on previously mined lands (i.e., conductivity ($\mu\text{S cm}^{-1}$)) were not different across treatment types (Table 1).

3.3.2. *Dipnet Surveys*

We detected nine amphibian species during dipnet surveys, with 693 captures in YFRA, 651 captures

in OFRA, 700 captures in REGEN, and 781 captures in MAT sites. The most commonly captured species was the Wood Frog (n = 1313 captures), followed by the Spotted Salamander (n = 368), Green Frog (n = 309), American Toad (*Anaxyrus americanus*; n = 296), Spring Peeper (*Pseudacris crucifer*; n = 224), Eastern Newt (n = 179), Gray Treefrog (n = 116), Four-toed Salamander (*Hemidactylium scutatum*; n = 19), and Pickerel Frog (*L. palustris*; n = 1). Most species were detected in all treatment types (Spotted Salamander, Four-toed Salamander, Green Frog, Wood Frog, Eastern Newt, and Spring Peeper); however American Toads were only detected in YFRA sites, Gray Treefrogs were detected in all treatment types except MAT, and the one Pickerel Frog was detected in OFRA. Overall, we documented eight species in created wetlands within the YFRA treatment, eight species in OFRA treatments, seven species in the REGEN treatment, and seven species in MAT treatment. We detected no federally protected amphibians or those considered a conservation priority by the state of West Virginia.

Table 3.1. ANOVA *p*-values, F-statistics, and average values of all site attributes and chemistry data collected in recently created wetlands on sites restored 2–5 years ago via the Forestry Reclamation Approach (YFRA), created wetlands in sites restored 8–11 years ago via the Forestry Reclamation Approach (OFRA), wetlands in naturally regenerated (REGEN) minelands (>40 years old), and wetlands in unmined, mature (MAT) forests. Superscript letters (i.e., a, b) represent the Tukey’s Honestly Significant Difference groupings.

Variable	<i>p</i> -Value	F-Stat	YFRA	OFRA	REGEN	MAT
Wetland Area (m ²)	0.729	0.436	57.99 ^a	71.63 ^a	81.66 ^a	49.66 ^a
Canopy Cover (%)	0.004	5.67	0 ^b	10.45 ^{ab}	53.68 ^a	54.75 ^a
Conductivity (μS cm ⁻¹)	0.316	1.23	44.4 ^a	52.09 ^a	23.24 ^a	33.39 ^a
pH (H ⁺)	0.001	6.71	6.74 ^a	6.78 ^a	5.77 ^b	6.15 ^{ab}
NO ₃ -N (mg L ⁻¹)	0.407	1	0.002 ^a	0 ^a	0 ^a	0 ^a
Turbidity (FTU)	0.328	1.2	8.57 ^a	4.02 ^a	6.35 ^a	2.88 ^a
TOC (mg L ⁻¹)	0.022	3.76	3.75 ^{ab}	4.31 ^a	2.73 ^b	3.19 ^{ab}
Ca (mg L ⁻¹)	0.47	0.87	4.77 ^a	6.78 ^a	1.78 ^a	5.14 ^a
Mg (mg L ⁻¹)	0.131	2.04	3.04 ^a	2.78 ^a	1.03 ^a	0.49 ^a
Fe (mg L ⁻¹)	0.397	1.02	1.74 ^a	3.9 ^a	7.81 ^a	1.74 ^a
NO ₂ -N (mg L ⁻¹)	0.179	1.76	0.29 ^a	0.2 ^a	0.42 ^a	0.39 ^a
Al (mg L ⁻¹)	0.054	2.87	0.15 ^a	0.1 ^a	0.68 ^a	0.3 ^a
Na (mg L ⁻¹)	0.505	0.8	0.75 ^a	0.56 ^a	0.55 ^a	0.83 ^a
Mn (mg L ⁻¹)	0.261	1.410	0.31 ^a	1.02 ^a	0.99 ^a	0.13 ^a
K (mg L ⁻¹)	0.117	2.15	1.39 ^a	1.39 ^a	0.76 ^a	0.67 ^a
Cl (mg L ⁻¹)	0.172	1.790	0.67 ^a	0.7 ^a	0.71 ^a	1.04 ^a
NH ₄ -N (mg L ⁻¹)	0.569	0.685	0.02 ^a	0.01 ^a	0.1 ^a	0.01 ^a
SO ₄ (mg L ⁻¹)	0.188	1.71	4.77 ^a	2.23 ^a	2.74 ^a	2.42 ^a

3.3.3. Occupancy and Species Richness

The mean amphibian occupancy for each treatment type with $\Psi_{YFRA} = 0.50$ (CI = 0.14, 0.86), $\Psi_{OFRA} = 0.60$ (CI = 0.22, 0.91), $\Psi_{REGEN} = 0.45$ (CI = 0.12, 0.83), and $\Psi_{MAT} = 0.45$ (CI = 0.15, 0.79)). The posterior distribution for wetland area (0.54 (CI = -0.07, 2.73)) indicated a mostly positive relationship, suggesting mean amphibian occupancy increased as wetland size increased. Mean amphibian detection was $p = 0.49$ (CI = 0.20, 0.78), and was not influenced by date ($\beta_1 = 0.15$ (CI = -0.13, 0.44)).

We found few effects of treatment type or wetland area on species-specific occupancy (Table 2), yet occupancy estimates varied among species (Table 3). For example, Spotted Salamanders had high occupancy estimates at all treatment types ($\Psi_{YFRA} = 0.79$ (CI = 0.49, 0.96), $\Psi_{OFRA} = 0.93$ (CI = 0.78, 0.99), $\Psi_{REGEN} = 0.821$ (CI = 0.56, 0.97), and $\Psi_{MAT} = 0.79$ (CI = 0.39, 0.98)). The occupancy of Eastern Newts was also relatively high across treatments ($\Psi_{YFRA} = 0.65$ (CI = 0.32, 0.92), $\Psi_{OFRA} = 0.87$ (CI = 0.66, 0.98), $\Psi_{REGEN} = 0.72$ (CI = 0.43, 0.94), and $\Psi_{MAT} = 0.62$ (CI = 0.19, 0.95)), whereas Wood Frogs occupancy estimates were relatively lower overall among treatments ($\Psi_{YFRA} = 0.42$ (CI = 0.15, 0.72), $\Psi_{OFRA} = 0.47$

(CI = 0.19, 0.76), $\Psi_{\text{REGEN}} = 0.51$ (CI = 0.23, 0.81), and $\Psi_{\text{MAT}} = 0.37$ (CI = 0.06, 0.80). Green Frog occupancy estimates had wide confidence intervals at all treatment types with $\Psi_{\text{YFRA}} = 0.50$ (CI = 0.19, 0.82), $\Psi_{\text{OFRA}} = 0.61$ (CI = 0.29, 0.89), $\Psi_{\text{REGEN}} = 0.56$ (CI = 0.28, 0.86), and $\Psi_{\text{MAT}} = 0.47$ (CI = 0.10, 0.89) and exhibited positive relationship to wetland area ($\alpha_4 = 0.97$ (CI = 0.20, 2.34)). For most species, date did not influence detection probability (Table 2). Mean species richness was similar across treatment type with $\text{SpRYFRA} = 4.21$ (CI = 3.13, 5.88), $\text{SpROFRA} = 5.11$ (CI = 4.00, 6.63), $\text{SpRREGEN} = 4.37$ (CI = 3.50, 5.75), and $\text{SpRMAT} = 3.82$ (CI = 2.88, 5.25).

Table 3.2. Mean parameter estimates (α) and 95% credible intervals for occupancy of nine pond-breeding amphibian species. Amphibians were sampled in recently created wetlands on sites restored via the Forestry Reclamation Approach (2–5-year old (YFRA) and 8–11-year (OFRA), wetlands in naturally regenerated (REGEN) minelands (>40 years old), and wetlands in unmined, mature (MAT) forests. Bold values indicate treatment types or covariates influencing amphibian occupancy and detection (i.e., 95% CIs that do not contain zero).

Species	YFRA	OFRA	REGEN	MAT	Wetland Area	Date
<i>Anaxyrus americanus</i>	0.90 (–0.76, 2.73)	0.24 (–2.12, 1.90)	–0.24 (–2.30, 1.35)	–3.38 (–6.48, –0.25)	–0.01 (–1.68, 0.87)	0.15 (–0.14, 0.44)
<i>Ambystoma maculatum</i>	–0.42 (–2.34, 1.21)	1.05 (–0.53, 2.71)	–0.17 (–1.89, 1.33)	1.68 (–1.15, 4.83)	0.63 (–0.18, 1.78)	0.19 (0.01, 0.58)
<i>Hemidactylium scutatum</i>	0.15 (–2.05, 2.32)	0.61 (–1.62, 2.50)	0.25 (–1.46, 2.23)	1.62 (–2.29, 5.93)	0.52 (–0.51, 1.76)	0.22 (–0.40, 0.67)
<i>Hyla versicolor</i>	0.93 (–0.62, 2.66)	1.09 (–0.34, 2.64)	–0.26 (–2.09, 1.18)	–1.42 (–4.29, 1.55)	0.51 (–0.24, 1.40)	0.12 (–0.12, 0.53)
<i>Lithobates clamitans</i>	–0.09 (–1.79, 1.40)	0.41 (–1.28, 1.87)	0.24 (–1.14, 1.80)	–0.13 (–2.8, 2.69)	0.97 (0.20, 2.34)	0.08 (–0.35, 0.42)
<i>Lithobates palustris</i>	0.02 (–2.32, 2.15)	0.81 (–1.18, 2.58)	–0.11 (–2.22, 1.74)	–1.05 (–6.28, 5.72)	0.53 (–0.80, 2.01)	0.18 (–0.33, 0.78)
<i>Lithobates sylvaticus</i>	0.08 (–1.46, 1.53)	0.30 (–1.37, 1.69)	0.46 (–0.88, 2.10)	–0.66 (–3.28, 1.99)	0.34 (–0.37, 1.00)	0.07 (–0.42, 0.44)
<i>Notophthalmus viridescens</i>	–0.14 (–1.89, 1.40)	1.29 (–0.20, 2.82)	0.20 (–1.23, 1.79)	0.63 (–2.08, 3.65)	0.73 (–0.02, 1.94)	0.2 (–0.13, 0.59)
<i>Pseudacris crucifer</i>	0.49 (–1.15, 2.30)	0.60 (–1.07, 2.08)	–0.22 (–2.06, 1.24)	–1.90 (–4.80, 0.97)	0.63 (–0.07, 1.53)	0.15 (–0.28, 0.59)

Table 3.3. Mean estimated occupancy (Ψ) and 95% credible intervals of nine amphibian species across four treatment types. Amphibians were sampled in recently created wetlands on sites restored via the Forestry Reclamation Approach (2–5-year old (YFRA) and 8–11-year (OFRA)), in wetlands in naturally regenerated (REGEN) minelands (>40 years old), and in wetlands within unmined, mature (MAT) forests.

Species	Ψ_{YFRA}	Ψ_{OFRA}	Ψ_{REGEN}	Ψ_{MAT}
<i>Anaxyrus americanus</i>	0.12 (0.02, 0.36)	0.08 (0.01, 0.25)	0.05 (0.003, 0.18)	0.06 (0.003, 0.25)
<i>Ambystoma maculatum</i>	0.79 (0.49, 0.96)	0.93 (0.79, 0.99)	0.82 (0.55, 0.97)	0.79 (0.39, 0.98)
<i>Hemidactylium scutatum</i>	0.79 (0.29, 0.99)	0.85 (0.36, 0.99)	0.83 (0.41, 0.99)	0.74 (0.18, 0.99)
<i>Hyla versicolor</i>	0.44 (0.15, 0.79)	0.48 (0.18, 0.82)	0.22 (0.04, 0.55)	0.24 (0.03, 0.70)
<i>Lithobates clamitans</i>	0.50 (0.19, 0.82)	0.61 (0.29, 0.89)	0.58 (0.28, 0.86)	0.47 (0.10, 0.89)
<i>Lithobates palustris</i>	0.38 (0.01, 0.99)	0.46 (0.02, 0.99)	0.37 (0.01, 0.99)	0.36 (0.004, 0.99)
<i>Lithobates sylvaticus</i>	0.42 (0.15, 0.72)	0.47 (0.19, 0.76)	0.51 (0.23, 0.81)	0.37 (0.06, 0.80)
<i>Notophthalmus viridescens</i>	0.65 (0.32, 0.92)	0.87 (0.66, 0.98)	0.72 (0.43, 0.94)	0.62 (0.19, 0.95)
<i>Pseudacris crucifer</i>	0.26 (0.05, 0.63)	0.28 (0.07, 0.58)	0.16 (0.02, 0.41)	0.18 (0.02, 0.57)

3.3.4. Abundance

Treatment type and wetland area had varying influence on amphibian abundance (Figures 3.2 and 3.3). The estimated abundance of Wood Frogs was positively associated with MAT ($\beta_1 = 3.80$, (CI = 3.62, 3.90)) and wetland area ($\beta_5 = 0.32$ (CI = 0.25, 0.38)) and negatively associated with YFRA ($\beta_2 = -2.66$ (CI = -3.13, -2.23)) and OFRA ($\beta_3 = -0.84$ (CI = -1.06, -0.63)). REGEN ($\beta_4 = -0.10$ (CI = -0.28, 0.08)) did not influence Wood Frog abundance. Wood Frog mean per-individual detection was $p = 0.44$ (95% CI = 0.42, 0.47), and detection was primarily negatively influenced by date ($\alpha_2 = -0.08$ (95% CI = -0.22, 0.05)). Abundance of Green Frogs was positively associated with MAT ($\beta_1 = 1.46$ (CI = 0.99, 1.88)) and wetland area ($\beta_5 = 1.08$ (CI = 0.94, 1.24)), and negatively associated with YFRA ($\beta_2 = -1.77$ (CI = -2.7, -0.93)) and OFRA ($\beta_3 = -0.67$ (CI = -1.3, -0.04)). The estimated abundance of Green Frogs was not affected by REGEN ($\beta_4 = 0.37$ (CI = -0.10, 0.86)). The mean per-individual detection probability for Green Frogs was $p = 0.28$ (95% CI = 0.19, 0.36), and detection was primarily positively influenced by date ($\alpha_2 = 0.35$ (95% CI = -0.03, 0.73)). The estimated abundance of Spotted Salamanders was positively associated with MAT ($\beta_1 = 2.41$ (CI = 2.06, 2.77)) and OFRA ($\beta_3 = 0.63$ (CI = 0.27, 0.99)), and negatively associated with YFRA ($\beta_2 = -0.43$ (CI = -0.88, -0.001)) and REGEN ($\beta_4 = -0.43$ (CI = -0.88, 0.00)). Mean per-individual detection probability for Spotted Salamanders was $p = 0.24$ (95% CI = 0.18, 0.30), and detection was marginally influenced by date ($\alpha_2 = 0.14$ (95% CI = -0.04, 0.34)). The abundance of Eastern Newts was positively associated with MAT ($\beta_1 = 1.35$ (CI = 0.65, 2.16)), OFRA ($\beta_3 = 1.12$ (CI = 0.53, 1.75)) and wetland area ($\beta_5 = 0.63$ (CI = 0.45, 0.81)). YFRA ($\beta_2 = -0.04$ (CI = -0.79, 0.71)) and REGEN ($\beta_4 = -0.28$ (CI = -0.98, 0.44)) did not influence Eastern Newt abundance. Mean per-individual detection probability for Eastern Newts was $p = 0.18$ (95% CI = 0.10, 0.30), and detection generally increased as date increased ($\alpha_2 = 0.27$ (95% CI = -0.03, 0.58)).

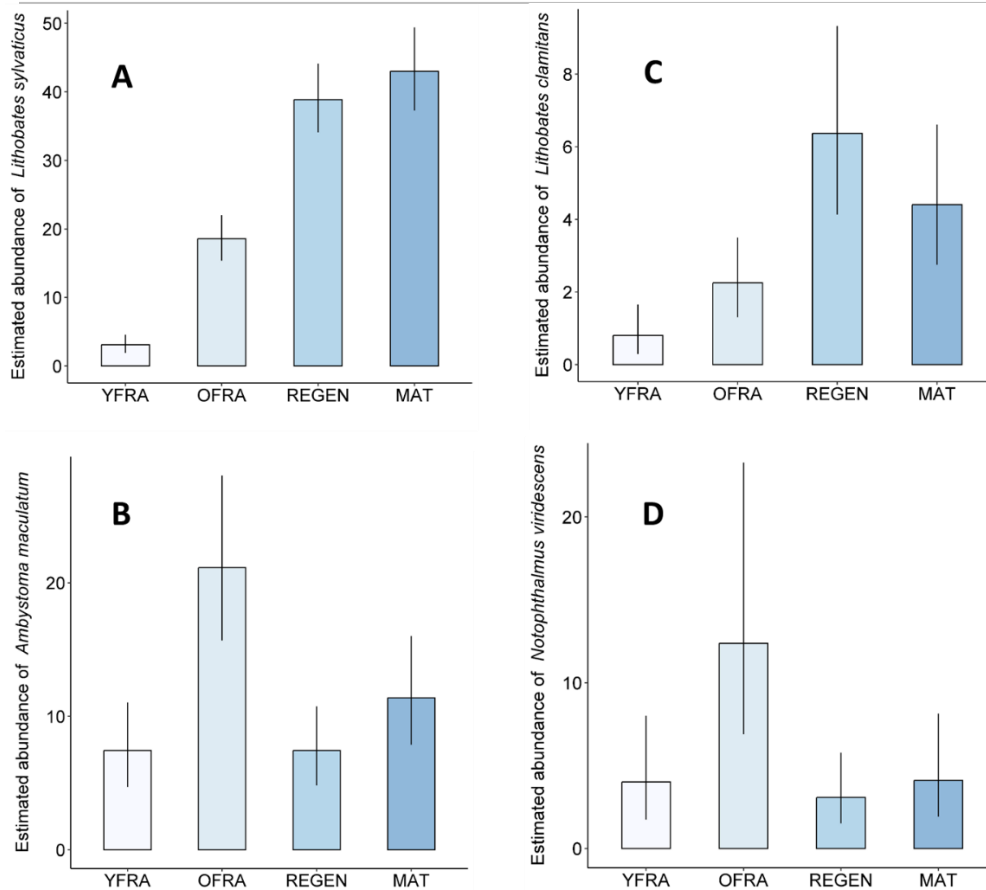


Figure 3.2. Estimated mean abundance (bars) of (A) Wood Frogs (*Lithobates sylvaticus*), (B) Spotted Salamanders (*Ambystoma maculatum*), (C) Green Frogs (*Lithobates clamitans*), and (D) Eastern Newts (*Notophthalmus viridescens*) in wetlands across four treatment types in the Monongahela National Forest (West Virginia USA). Amphibian count data were collected at 2–5-year old sites restored via the Forestry Reclamation Approach (YFRA), 8–11-year old sites restored via the Forestry Reclamation Approach (OFRA), wetlands in naturally regenerated (REGEN) minelands (>40 years old), and wetlands in unmined, mature (MAT) forests. Error bars indicate 95% credible intervals.

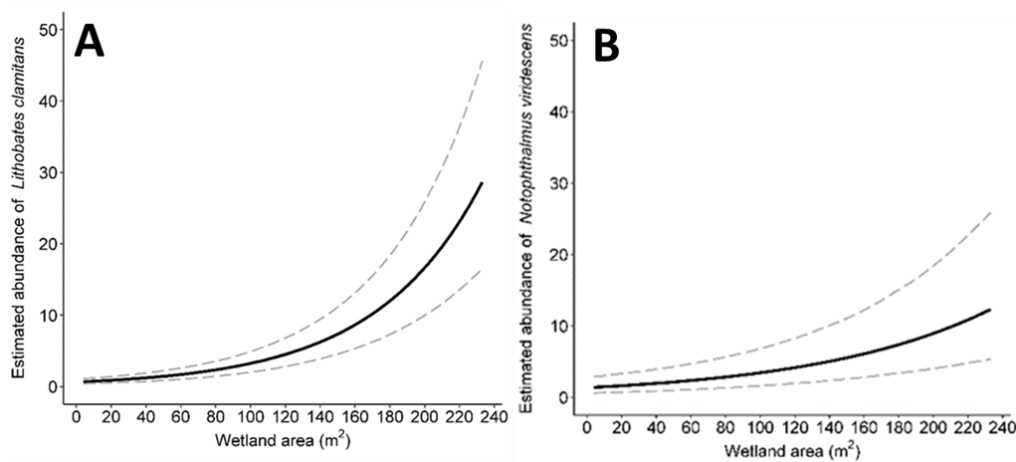


Figure 3.3. Effects of wetland area (m²) on estimated mean abundance (solid lines) of (A) Green Frogs (*Lithobates clamitans*) and (B) Eastern Newts (*Notophthalmus viridescens*) in created wetlands on sites restored via the Forestry Reclamation Approach, wetlands in naturally regenerated minelands (>40 years old), and wetlands in unmined, mature forests and natural wetlands in the Monongahela National Forest (West Virginia USA). Dashed lines indicate 95% credible intervals.

3.4. Discussion

The reforestation of surface mines and wetland creation provides habitat for multiple pond-breeding amphibian species in our study area. Few differences in wetland attributes were identified between created wetlands at sites restored via the FRA compared to wetlands on naturally regenerating mine lands (REGEN) and in mature, unmined forests (MAT). Amphibian species occupancy and richness also did not differ across treatment types; however, for amphibian abundance, treatment and wetland area effects varied by species. Estimated abundances for most species exhibited a positive association with MAT sites and wetland area, and a negative association with YFRA (except Eastern Newts). However Spotted Salamander and Eastern Newt abundances had positive associations with OFRA sites and were most abundant in these treatment types..

In general, the wetland attribute analyses showed similar physical conditions and water chemistry between constructed wetlands and naturally occurring wetlands in our study area. We found significant differences in canopy cover between treatments, with YFRA sites having the least canopy due to being replanted 2–5 years prior to this study. Only two water chemistry variables were different across treatments (i.e., TOC, pH), and these values were within a normal range for created and natural wetlands in our study area (Lambert et al. 2021). The lower mean pH in REGEN and MAT sites is likely due to the deposition of acidic conifer needles from planted Norway spruce in REGEN sites or red spruce in MAT sites (Hughes 2018), and from decades of acid deposition that have lowered soil and water pHs across the Monongahela National Forest (Elias et al. 2009). The slightly higher pH at FRA sites could also be attributed to the nature of some of the freshly plowed spoils exposing unweathered rock that can leach ions when exposed to water; this has been shown to exhibit circumneutral pH levels (Agouridis et al. 2012). We found that OFRA sites had significantly higher levels of TOC than REGEN sites. This result was unexpected as REGEN sites have higher percentages of canopy cover and elevated TOC levels are often the result of decaying plant matter (Ahn and Jones 2013). Site preparation techniques may explain these differences in TOC, as logs were placed in created wetlands during the restoration process, and wetland edges were planted with native wetland shrubs and plants. The logs originated from downed non-native conifers in the project area. While logs were used in both YFRA and OFRA sites, logs in the YFRA were relatively green when sampled whereas those in the OFRA showed considerable decomposition which likely contributed to the elevated TOC. Across FRA treatments, we found marginally significant differences in mean concentrations of Al, an element which is toxic to amphibians at low pH values (Freda 1986; Freda 1991). However, mean Al concentrations did not exceed the *Lithobates* mean chronic value (>10,684 µg/L) (US EPA 2018) and Al is insoluble in water with pH values > 5.5 (Lindsay and Walthall 2020), which is lower than the pH values found at our sites. Finally, we noted that conductivity did not differ across treatments. Conductivity is often elevated in aquatic systems on previously mined lands (Lindberg et al. 2011), and elevated conductivity is correlated with reductions in stream-inhabiting amphibian

occupancy and abundance (Hutton et al. 2020). Our study wetlands had mean conductivity levels substantially lower than wetlands on reclaimed minelands in Kansas (USA) (Buckardt 2023), and we found that conductivity levels were well below the U.S. Environmental Protection Agency conductivity benchmark of $300 \mu\text{S cm}^{-1}$ for aquatic life in the Central Appalachian region (US EPA 2011).

We found that pond-breeding amphibian species occupancy and richness did not differ among treatment types. Although relatively few studies have been conducted on pond-breeding amphibian occupancy and richness in post-mining landscapes, our results generally support previous findings. For example, Stiles et al. (2017) found that pond-breeding amphibian communities inhabiting a reclaimed surface mine in Indiana (USA) had over 15,000 individuals belonging to 14 species; the diversity and abundance they attributed to the presence of a variety of wetland types and terrestrial habitat that allowed successful colonization and reproductive success. Pond-breeding amphibian community composition and species occupancy was also found to be comparable among wetlands regardless of mining history in Kansas (USA) (Buckardt et al. 2023). Finally, a previous investigation examined pond-breeding amphibian occupancy across four wetland age classes (2, 4, 6, 8 years post-construction) in our study area and found that high mean occupancy rates for Spotted Salamanders (i.e., ~ 0.7 across all age classes) and occupancy rates for Green Frogs in 4-year-old wetlands were ~ 1.00 (Lambert et al. 2021). Overall, the lack of influence by treatment type indicates that constructed wetlands on FRA sites supported similar communities to wetlands on naturally regenerating mine lands (REGEN) and wetlands in mature forests (MAT) in our study area. Collectively, these studies and our own indicate restoration creates habitat for amphibians, which might increase resiliency to climate change through increased habitat availability.

Despite similarities in occupancy and community composition, we found that pond-breeding amphibian abundances varied across treatment types. Estimated abundances of all four focal species were positively associated with MAT sites, which was expected as MAT sites did not experience past mining activities. In addition, estimated abundance of Wood Frogs and Green Frogs were reduced at YFRA and OFRA sites compared to MAT sites, whereas estimated abundances of Spotted Salamanders and Eastern Newts were greatest in OFRA sites. Several mechanisms may explain the variability in abundances. First, older wetlands allow more time for species colonization, which could lead to increased abundances in older wetlands (Laan and Verboom 1990). We found estimated abundances of most species were negatively associated with YFRA and positively associated with MAT treatment. However, if wetland age was an important factor determining abundance, we would have expected to find an effect of REGEN as wetlands within this treatment were on mined lands dating to the pre-SMCRA (i.e., pre-1977) era. Furthermore, Eastern Newts and Spotted Salamanders exhibited greatest estimated abundances in OFRA treatment, with wetland ages ranging from 8–11 yrs. Thus, a second explanation may relate to predation by Spotted Salamander larvae and Eastern Newts on tadpoles. Spotted Salamander larvae prey on a wide variety of anuran tadpoles (Resetarits and Wilbur 1989), and Eastern Newts are known to limit Wood Frog reproductive success via consumption of Wood Frog eggs (Kross and Richter 2016). Both Spotted Salamanders and Eastern Newts were most abundant in OFRA treatments, and it is possible that these salamander species may lead to reductions in some anurans in this treatment type. Finally, we note that abundances of pond-breeding amphibians exhibit significant inter-annual variability (Marsh 2001), and therefore monitoring that extends beyond a single sampling season may be necessary to elucidate the drivers of abundance.

The covariate with the most consistent influence on species occupancy and abundance was wetland area. Wetland area had a positive effect on Green Frog occupancy and abundance and Wood Frog and Eastern Newt abundance. Previous studies have also highlighted the relationship between larger wetlands and increased occupancy rates and abundance (Lambert et al. 2021; Drayer et al. 2020; Semlitsch et al. 2015). Specifically, seasonal wetlands that range from 1000–10,000 m^2 support the greatest number of species (Semlitsch et al. 2015; Babbitt 2005), because the larger wetland size leads to decreased inter- and intra-species competition and overall stress (Millikin et al. 2019). The wetlands at our study site ranged from 4.7 m^2 to 233.2 m^2 , suggesting that creating larger seasonal wetlands during restoration activities may promote higher occupancy rates and greater amphibian abundances.

We acknowledge that several unmeasured environmental attributes may partially explain patterns of pond-breeding amphibian occupancy and abundances. For instance, wetland vegetation cover has been shown to negatively influence occupancy rates of Spotted Salamanders and Green Frogs, while positively influencing the abundance of Eastern Newts (Lambert et al. 2021). Wetland vegetation serves as important egg deposition

locations and offers larvae cover from predators, but the decomposition of wetland plants can reduce oxygen levels in wetlands and/or produce phytochemicals that increase larval mortality and decrease growth rates (reviewed in Burrow and Maerz 2022). Landscape and spatial configuration of created wetlands can also influence amphibian occupancy and abundance, as wetlands constructed close to existing habitat may lead to higher colonization rates whereas spatially isolated wetlands often have lower colonization rates (Sawatzky et al. 2019). Our reclamation activities occurred within national forest boundaries, and this likely aided colonization rates for amphibians. Nevertheless, reclamation techniques used at our study sites, such as woody debris loading and the creation of over 800 wetlands, likely provided terrestrial refugia, promoted habitat connectivity, and facilitated colonization of created wetlands.

3.5. Conclusions

Our results indicate that the FRA coupled with wetland creation restores habitat for pond-breeding amphibians within a relatively short timeframe (<10 yrs). The strong amphibian community response at our study sites was likely due to: (1) the high density of created, seasonal wetlands (>800) resulting in more breeding opportunities; and (2) site preparation techniques, including soil decompaction, woody debris loading, and tree planting, that (a) promote microhabitats and microclimate refugia for amphibians during the non-breeding season, and (b) facilitate movement (i.e., dispersal, migration) that leads to successful colonization of newly created wetlands.

3.6. References

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Section 4. Response of Terrestrial Salamanders to the Forestry Reclamation Approach on Legacy Surface Mines

4.1. Introduction

Coal mining has impacted land cover in the Appalachian region. The Surface Mining Control and Reclamation Act (SMCRA) of 1977 requires U.S. coal mining companies to reclaim mined lands to avert soil erosion and landslides. Traditional reclamation practices involve soil compaction and planting of non-native grasses to form grasslands (Angel et al. 2015). Traditional approaches result in loss of topographic complexity, poor hydrological function, and limited root growth, which constrains reforestation (Burger et al. 2013). Recently, practitioners restored some traditionally reclaimed sites using an adaptation of the Forestry Reclamation Approach (FRA) for legacy surface mines (Burger et al. 2017). Restoration techniques included soil de-compaction, non-native species removal, planting of native trees, woody debris loading, and wetland creation to accelerate forest succession and provide environmental services, including wildlife habitat (Lambert et al. 2021).

Terrestrial salamanders (Family Plethodontidae) are abundant in Appalachian forests, but have low abundances on traditionally reclaimed minelands (Wood and Williams 2013). However, the response of terrestrial salamanders to the FRA has not been investigated. For this study, we selected 30 sites on legacy minelands and mature, red spruce (*Picea rubens*) - northern hardwood forests within/around Mower Tract and Sharp's Knob (Randolph and Pocahontas Counties) in the Monongahela National Forest (MNF) in West Virginia (USA). In addition, we selected 24 sites in the Cumberland Plateau in Kentucky (USA). Due to low captures in Kentucky (see results below), we focused our analysis on West Virginia captures. Specifically, we estimated the abundance of Red-backed Salamanders (*Plethodon cinereus*) across restored legacy mines of two FRA age classes [i.e., young FRA (2-5 yrs since planting; YFRA) and older FRA (8-11 yrs since planting; OFRA)], and compared abundances to naturally regenerating forests on pre-SMCRA minelands (> 40 yrs since mining; REGEN) and mature, unmined forests (MAT). We hypothesized that salamander would be least abundant in YFRA sites, and abundances would increase across the chronosequence of forest age (i.e., YFRA < OFRA < REGEN < MAT).

4.2. Methods

Salamander surveys were conducted in YFRA (n = 8), OFRA (n = 8), REGEN (n = 7), or MAT (n = 7) treatments in WV. YFRA sites were restored between 2017 and 2020 and were loaded with woody debris prior to planting of *P. rubens* and 12 hardwood species. YFRA sites were characterized by extensive herbaceous groundcover; seedlings had a mean height of < 1 m. OFRA sites were restored in 2011 and 2014, and planted with seedlings of *P. rubens* and 3 hardwood species. Sapling height at OFRA sites ranged from 2-4 m.

REGEN sites were mined prior to the implementation of SMCRA and initially reclaimed as grasslands; however, planting of non-native conifers (Norway spruce; *P. abies*) and moderate recolonization by native trees led to partial to full canopy closure at some REGEN sites (Branduzzi 2020). MAT sites were unmined, second-growth forest. All sites were at least 200 m apart to promote independence. In October 2021, we placed 10 pine coverboards (measuring 30 x 30 x 2.5 cm) in a 1 x 10 m array at each study site. We surveyed boards on six occasions at each site from late May through early July 2022. Salamander surveys involved searching under each board within an array, and recording the species and the total number of individuals detected. Count data were generated by summing salamander captures under 10 boards at each site.



Figure 4.1. Red-backed Salamander (*Plethodon cinereus*) abundance was assessed in legacy surface mines restored via the Forestry Reclamation Approach, unrestored legacy surface mines, and mature, unmined forests in the Monongahela National Forest (West Virginia). Photo Credit: Steven J. Price

We used a Bayesian binomial mixture model (Royle 2004) to examine effects of treatment type on salamander abundance, following the approach outlined in Drayer et al. (2020). Using count data from repeat visits, we modeled abundance (λ) with a Poisson distribution. Heterogeneity in abundance among populations was modelled using a Poisson-regression formulation of mean abundances, given by $\log(\lambda_i) = \beta_0 + \beta_1 \text{YFRA} + \beta_2 \text{OFRA} + \beta_3 \text{REGEN}$. MAT served as the reference category (β_0). Heterogeneity in detection (p) was identified by modelling associations between date (scaled with mean value of zero), and p_i such that $\text{logit}(p_{ij}) = \alpha_0 + \alpha_1 \text{date}_{ij}$. The model used uninformative priors; we assumed $\beta_0, \beta_1, \beta_2, \beta_3 \sim N(0, 10^2)$, $\alpha_0 \sim N(0, 1.6^2)$ and $\alpha_1 \sim N(0, 10^2)$. Posterior summaries for each parameter were based on 200,000 Markov chain Monte Carlo iterations with a 100,000 sample burn-in and a thinning rate of 50 providing a total of 6,000 samples from which we approximated posterior summary statistics for each model parameter. We inferred significance for treatment parameter if the 95% credible intervals did not contain zero. Analyses were conducted in WinBUGS (V 1.4; Spiegelhalter et al., 2003) with data handling in R (V 4.2.2.). For all monitored parameters, the Gelman-Rubin diagnostic was at or below 1.02 indicating convergence (Gelman and Rubin 1992).

4.3. Results

The majority of captures ($n=33$; 60%) were of the red-backed salamander (*Plethodon cinereus*). Other species detected included Eastern Newt (*Notophthalmus viridescens*; $n=2$), Northern Two-lined Salamander (*Eurycea bislineata*; $n=7$), and the Allegheny Mountain Dusky Salamander (*Desmognathus ochrophaeus*; $n=13$). In Kentucky, we documented a total of 3 captures: Eastern Newt ($n=1$), Northern Slimy Salamander (*Plethodon glutinosus*; $n=1$); Long-tailed Salamander (*Eurycea longicauda*; $n=1$).

Counts of *P. cinereus* across treatments included two in YFRA, zero in OFRA, 31 in REGEN, and 19 in MAT. Estimated abundance of *P. cinereus* was negatively associated with YFRA ($\beta_1 = -2.21$ [95% CI = -4.17 – -1.62]) and OFRA ($\beta_2 = -9.91$ [95% CI = -23.53 – -2.55]). Credible intervals overlapped zero for the REGEN treatment ($\beta_3 = 0.22$ [95% CI = -0.647 – 1.09]) and the MAT treatment ($\beta_0 = 0.641$ [95% CI = -0.1 – 1.23]), however, the majority of the distribution for MAT was positive. Estimated abundance was 0.27 (95% CI = 0.3 – 0.78) individuals per 10 coverboards in YFRA sites, 0.02 (95% CI = 0 – 0.14) in OFRA sites, 2.5 (95% CI 1.18 – 4.43) in REGEN sites and 2.02 (95% CI = 0.9 – 3.67) in MAT sites (Figure 2). Mean per-individual detection probability was 0.29 (95% CI = 0.18 – 0.422), and negatively associated with date ($\alpha_1 = -0.56$ (95% CI = -0.93 – -0.2).

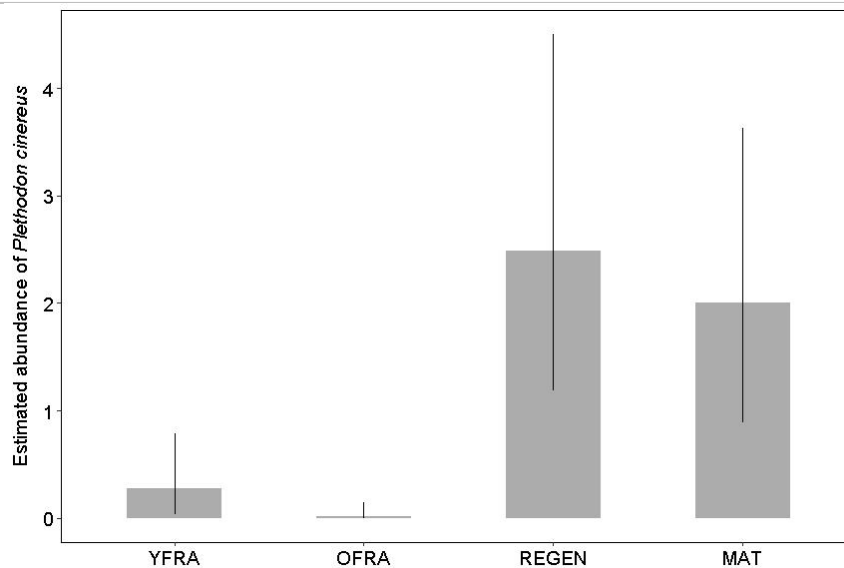


Figure 4. 2. Estimated mean abundance of Red-backed Salamanders (*Plethodon cinereus*) within 1 x 10 m coverboard arrays in young Forestry Reclamation Approach (FRA) forests (2-5 yrs since planting; YFRA), older FRA forests (8-11 yrs since planting; OFRA), naturally regenerating (REGEN) forests on minelands, and unmined mature (MAT) forests in the Monongahela National Forest (West Virginia). Error bars indicated 95% credible intervals.

4.4. Discussion and Conclusions

Our study provides the first assessment of terrestrial salamanders responses to FRA. We found that *P. cinereus* abundances were low within FRA sites. Terrestrial salamanders respire through their skin; consequently, cool temperatures and moist conditions are required. In forests, cool temperatures are positively associated with high levels of canopy cover, and high moisture levels are maintained by woody debris and thick layers of leaf litter (Chen et al. 1999). Thus, the FRA sites may not yet provide the conditions preferred by salamanders. Although the REGEN parameter estimate had no overall effect on abundance, we consider REGEN sites a model for understanding the potential long-term outcomes of FRA. The majority of *P. cinereus* detections were at REGEN sites with closed canopy due to the planting of non-native conifers. If the FRA results in accelerated forest succession and canopy closure as seen in previous FRA studies (Groninger et al. 2007), we would expect salamander abundances to increase as forests mature. Salamander populations often take decades to recover from large-scale disturbances in forests, with populations generally increasing with forest regeneration time and concomitant increases in canopy cover (Tilghman et al. 2012). Recovery of terrestrial salamander populations is often slow as most species exhibit high site fidelity, small home ranges, and limited dispersal abilities (Kleeberger and Werner 1982, Gibbs 1998). However, colonization of unoccupied sites can occur rapidly if mature forest occurs in close proximity (i.e., 5 - 25 m) to restored habitat patches that have moisture, canopy coverage, and leaf litter suitable for salamanders (Marsh et. al 2004). Thus, managers should consider the proximity of restoration activities in relation to mature forests if recovery of terrestrial salamanders is a priority. We also suggest long-term monitoring to fully assess the response of salamanders to FRA.

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Section 5. Mammals Detected via Automated Recording Units and Camera Traps on Reforested Surface Mines in West Virginia and Kentucky

5.1. Introduction

Surface mine reclamation in the U.S. is regulated under the Surface Mine Control and Reclamation Act (SMCRA) of 1977, part of which was aimed at reducing the threat of erosion and landslides from surface-mined sites throughout the steep sloped Appalachian region. Reclamation under SMCRA, through years of interpretation and enforcement, eventually matured into what is now considered traditional or conventional surface mine reclamation in the region, characterized by heavy minesoil compaction and final land-uses of pasture/grassland or wildlife habitat (grassland/shrublands) (Angel et al. 2015). Traditionally reclaimed mines (grassland/shrubland) have been shown to provide habitat for wildlife generalist species, such as bobwhite quail, turkey, rabbits, deer and elk, yet forest-dependent wildlife are often missing from traditionally reclaimed minelands (Cox and Maehr 2005, Larkin et al. 2008; Becker et al. 2015).

Recently, reclamation practitioners have restored some traditionally reclaimed sites (i.e., grassland/shrubland) using an adaptation of the Forestry Reclamation Approach (FRA) for legacy surface mines (Burger et al. 2017). The FRA utilizes soil de-compaction, non-native species removal, planting of native trees, woody debris loading, and wetland creation to accelerate forest succession and provide environmental services. Although FRA has been demonstrated to be successful at reforestation across the Appalachian region, until recently very little is known about how FRA techniques influence habitat required by many forest-dependent wildlife species (see Lambert et al. 2021; Sherman et al. 2024; but see Price et al. 2024).

The central Appalachian region is well known for mammal diversity, with some species (West Virginia Northern Flying Squirrel (*Glaucomys sabrinus fuscus*), Spotted Skunk (*Spilogale putorius*) and Southern Water Shrew (*Sorex palustris punctulatus*)) considered to be state or regionally imperiled. In addition, some bat species such as the Indiana bat (*Myotis sodalist*), northern long-eared bat (*Myotis septentrionalis*), Rafinesque's big-eared bat (*Corynorhinus rafinesquii*) and Tri-colored bat (*Perimyotis subflavus*) are being considered for federal listing under the Endangered Species Act of 1973. In this study, we examined the mammal community composition and abundances via automated recording units (ARU) and camera trap surveys across restored legacy mines of two FRA age classes [i.e., young FRA (1-5 yrs since planting; YFRA) and older FRA (8-23 yrs since planting; OFRA)], and compared abundances to mature, unmined forests (MAT) in Kentucky and West Virginia. Our primary goal is to provide a checklist of mammal species in both regions.

5.2. Methods

5.2.1. Study Sites, Sampling, and Analysis

Automated recording units were deployed at 12 sites (4 YFRA, 4 OFRA, 4 MAT) in West Virginia, and 12 sites in Kentucky (4YFRA, 4 OFRA, 4 MAT) (See Figure 2.1; 6.1). Bats were detected using Song Meter SM3BAT detectors with external SMM-U2 ultrasonic microphones (Wildlife Acoustics, Maynard, MA). We examined the acoustic recordings using Kaleidoscope Pro 5 (Wildlife Acoustics, Maynard, MA). We appraised all files for bat vocalizations and removed files without any bat echolocation pulses, as they represented captures of ultrasonic waves from non-bat sources. Because individual recordings could have between 1 and >100 pulses, each recording varied in the amount of time and energy expended by the vocalizing bat. To compensate, we used Kaleidoscope to count individual echolocation pulses automatically in each recording. See Section 2 for details on deployment, activity definition and species identification. Finally, we manually counted feeding buzzes to evaluate foraging activity. Feeding buzzes are foraging attempts identified as a series of pulses emitted in quick succession (Russo et al. 2018). Recordings with ≥ 5 pulses were identified to species using the software's bat call reference library and specifying Bats of North America 5.4.0 and West Virginia or Kentucky as the region. See Section 2.2 for more details.

A modified version of this section in preparation for submission to Reclamation Sciences. Price, S.J., B. Snyder, L. Sherman, V. Burgess, J.L. Larkin, J. Cox, and C. Barton. In preparation. Mammals detected via automated recording units and camera traps on reforested surface mines in West Virginia and Kentucky (USA).

We deployed motion sensitive camera traps (BTC 6HDX Dark Ops 940; Browning Trail Cameras, Birmingham, AL, USA) to detect mammals at 12 sites in Kentucky and 12 in West Virginia (See Figure 2.1; Figure 6.1). Traps were deployed from March – July in West Virginia, and February through August in Kentucky. We mounted camera on 1.5-m tall stakes positioned ~1 m from wetlands. Consequently, the camera detection zone was approximately 2–4 m², allowing us to see the wetland and a small extent of the surrounding terrestrial border (Figure I.2). To avoid camera sensitivity bias, we set all cameras to motion activation via an infrared sensor with high motion sensitivity and programmed them to take a series of 3 high-definition photographs at a trigger speed of 0.2 seconds, following methods and procedures from Rovero et al. (2013). We identified each captured image to species by manually processing the images through Camelot Software package (<https://camelotproject.org/>).

5.3. Results and Discussion

5.3.1. Bat Surveys

We recorded for 12 full nights at each site in West Virginia, which resulted in a total of 12,110 records that contain bat calls. Of the 12,110 recordings, 6,282 sequences were identified to species. The following species were identified throughout the study area in decreasing order of occurrence: Red bat (*Lasiurus borealis*), Big brown bat (*Eptesicus fuscus*), Hoary bat (*Lasiurus cinereus*), Tri-colored bat (*Perimyotis subflavus*), Silver-haired bat (*Lasionycteris noctivagans*), and *Myotis* spp (Small-footed myotis – *Myotis leibii*; Little brown bat – *Myotis lucifugus*; Northern long-eared bat – *Myotis septentrionalis*; Indiana bat – *Myotis sodalis*). OFRA sites contained 6 species, followed by MAT sites with 5 species, and YFRA with four species; however the number of call sequences was greatest at YFRA sites (Table 5.1).

The bat surveys in Kentucky were conducted for three nights at each site, resulting in 8,319 records that contain bat calls. Of these recordings, 3,988 sequences were identified to species. *Myotis* species are difficult for the auto-identification software to differentiate, so the *Myotis* identifications were grouped together. The following species were identified in decreasing order of occurrence: Big brown bat (*Eptesicus fuscus*), Red bat (*Lasiurus borealis*), Hoary bat (*Lasiurus cinereus*), *Myotis* spp. (Gray bat – *Myotis grisescens*, Small-footed myotis – *Myotis leibii*, Little brown bat – *Myotis lucifugus*, Northern long-eared bat – *Myotis septentrionalis*, Indiana bat – *Myotis sodalis*), Tricolored bat (*Perimyotis subflavus*), Silver haired bat (*Lasionycteris noctivagans*), and Virginia big-eared bat (*Corynorhinus townsendii virginianus*). Although MAT sites contained the most species, the number of call sequences were greatest in the FRA treatments (Table 5.2).

Table 5.1. Number of call sequences (≥ 5 pulses) for each bat species listed by treatment type in the Monongahela National Forest (West Virginia). *Myotis* spp. include *M. leibii*, *M. lucifugus*, *M. septentrionalis*, and *M. sodalis*. YFRA sites were restored between 2017 and 2020, OFRA sites were restored in 2011 and 2014, and MAT sites were mature, unmined forests.

Species	Number of call sequences			Total
	YFRA	OFRA	MAT	
<i>Eptesicus fuscus</i>	1388	37	91	1669
<i>Lasiurus borealis</i>	820	318	522	2844
<i>Lasiurus cinereus</i>	311	232	187	1330
<i>Lasionycteris noctivagans</i>	0	66	73	139
<i>Myotis</i> spp.	5	5	0	44
<i>Perimyotis subflavus</i>	0	65	45	256
Total	2524	723	918	6282

Table 5. 2. Number of call sequences (≥ 5 pulses) for each bat species listed by treatment type in the Southeast Kentucky. *Myotis* spp. include *M. leibii*, *M. lucifugus*, *M. septentrionalis*, and *M. sodalis*. YFRA sites were restored in 2022, OFRA sites were restored in 2000 and 2016, and MAT sites were mature, unmined forests.

Species	Number of call sequences			Total
	YFRA	OFRA	MAT	
<i>Eptesicus fuscus</i>	252	1219	3	1474
<i>Lasiurus borealis</i>	715	93	68	876
<i>Lasiurus cinereus</i>	158	521	18	697
<i>Myotis</i> spp	94	12	72	178
<i>Perimyotis subflavus</i>	0	23	0	23
<i>Lasionycteris noctivagans</i>	0	0	13	13
<i>Corynorhinus townsendii</i>	0	0	12	12
Total	1219	1868	186	3273

5.3.2. Camera Trap Surveys

We processed 602,535 images from West Virginia study sites. We detected 9 mammal species in total with 4 species detected in the YFRA treatment, 6 in the OFRA treatment, and 7 in the MAT treatment (Table 5.3). White-tailed Deer (*Odocoileus virginianus*) were the most common species detected (185 detections), with similar number of detections across treatments (YFRA = 54, OFRA= 68; MAT = 63).

We processed 627,630 images from Kentucky study sites and detected 13 species. We detected 8 species in the YFRA treatment, nine species in the OFRA treatment, and 9 species in the MAT treatment (Table 5.4). White-tailed Deer were the most common species detected (589 total detections), followed by Elk (270 detections), and Eastern Gray Squirrel (258 detections; MAT only).

Table 5. 3. Mammals detected via camera traps in the Monongahela National Forest (West Virginia). YFRA sites were restored between 2017 and 2020, OFRA sites were restored in 2011 and 2014, and MAT sites were mature, unmined forests.

Species	Number of detections			
	YFRA	OFRA	MAT	Total
<i>Canis latrans</i>	1	1	7	9
<i>Didelphis virginiana</i>	0	4	2	6
<i>Lynx rufus</i>	0	0	1	1
<i>Odocoileus virginianus</i>	54	68	63	185
<i>Procyon lotor</i>	1	0	16	17
<i>Sylvilagus floridanus/Lepus americanus</i>	4	1	0	5
<i>Ursus americanus</i>	0	1	2	3
<i>Taiasciurus hudsonicus</i>	0	5	0	5
<i>Pekania pennanti</i>	0	0	2	2
Total	60	80	93	233

Table 5. 4. Mammals detected via camera traps in southeastern Kentucky. YFRA sites were restored in 2022, OFRA sites were restored in 2000 and 2016, and MAT sites were mature, unmined forests.

Species	Number of detections			
	YFRA	OFRA	MAT	Total
<i>Canis latrans</i>	10	6	10	26
<i>Bos taurus</i>	0	13	0	13
<i>Cervus canadensis</i>	229	41	0	270
<i>Didelphis virginiana</i>	0	0	8	8
<i>Glaucomy volans</i>	0	0	1	1
<i>Lynx rufus</i>	2	7	0	9
<i>Mustela vison</i>	0	0	1	1
<i>Odocoileus virginianus</i>	167	183	239	589
<i>Procyon lotor</i>	7	7	126	140
<i>Sciurus carolinensis</i>	0	0	258	258
<i>Sylvilagus floridanus</i>	23	10	2	35
<i>Vulpes vulpes</i>	0	0	1	1
<i>Ursus americanus</i>	1	2	0	3
Total	439	269	646	1354

This preliminary investigation is the first to document the community composition of mammals associated with wetlands on reclaimed surface mines in the Central Appalachian region. Our results indicate that bats are using reforested mined lands as habitat, including utilizing created wetlands as foraging locations. Although most bats detected are considered common species, the Tri-colored Bat is currently petitioned for inclusion on the US Endangered Species list and although we are unable to identify *Myotis* species, it is quite possible recordings could be gray bat or northern long-eared bat. Both of these species, as well as Virginia big-eared bat, are federally endangered. Red Bat, Hoary Bat, Silver-haired Bat, and Tri-colored Bat are considered priority species in West Virginia (West Virginia State Wildlife Action Plan; <https://wvdnr.gov/state-wildlife-action-plan/>), and the Tricolored Bat and Virginia Big-eared Bat considered priority species in Kentucky (Kentucky State Wildlife Action Plan; <https://app.fw.ky.gov/kyswap/>)

Terrestrial mammal diversity was greatest in MAT treatments compared to the FRA treatments. We did not detect any species currently considered for federal listing or priorities at the State-level. Notable species detected in West Virginia sites included Bobcat (*Lynx rufus*) (MAT only; 1 detection), Fisher (*Pekania pennanti*) (MAT only; 2 detections) and American Black Bear (*Ursus americanus*) (OFRA = 1 detection; MAT = 2 detections). In Kentucky, 3 Bobcats and 3 American Black Bears were detected; both species were detected in YFRA or OFRA treatments.

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Section 6. Restored Wetlands on Reforested Surface Mines Provide Breeding Habitat for Anurans in the Central Appalachians

6.1. Introduction

Human-induced land use changes are a major threat to biodiversity (Foley et al. 2005). Surface coal mining is a main driver of land use change, particularly in the Central Appalachians (USA) (Zipper 2011). Surface coal mining results in changes to topography, soil compaction, changes to local hydrology, introduction of non-native vegetation, and forest fragmentation (Wickham et al. 2013). Reclamation efforts attempt to mitigate some of the negative impacts of surface coal mining on the environment. With the passing of the Surface Mining Control and Reclamation Act (SMCRA) of 1977, coal mining companies are required to restore mined lands original topography and plant vegetation to prevent soil erosion and landslides (Office of Surface Mining 1977). Mined lands are most often reclaimed as grasslands; however, grassland reclaimed sites prevented the recolonization of tree species, which has resulted in the loss of forest-associated biodiversity (USEPA 2003; Bernhard and Palmer 2011; Wickham et al. 2013; Burger et al. 2013; Angel et al. 2015). Recently, the Forestry Reclamation Approach (FRA) has been used to restore forests on legacy surface mines originally reclaimed as grasslands. The FRA emphasizes non-native vegetation removal, soil decompaction, and planting of native trees and shrubs (Burger et al. 2013). At some locations, the FRA also utilizes woody debris loading and wetland creation (Lambert et al. 2021).

Wetland creation and reforestation of legacy surface mines offers the potential to benefit many wildlife species (Stiles et al. 2017; Lambert et al. 2021; Snyder et al. 2024). In particular, pond-breeding amphibians may benefit from the creation of new breeding habitats, and reforestation of terrestrial environmental may provide favorable microhabitat conditions. Indeed, larval and aquatic adult amphibians within wetlands on legacy mines often have high occupancy rates and abundances equal to or greater than those found in naturally occurring wetlands (Lambert et al. 2021; Buckardt et al. 2023; Sherman et al. 2024). One technique to monitor amphibian use of created wetlands involves the use of programmable audio recording units with microphones (automated recording units (ARU)). Anuran vocalizations collected via ARUs provides time-series data that results in little disturbance to the sampling area, which can enhance species detection (Digby et al. 2013). Furthermore, data acquired via ARUs can be particularly useful to detect rare species, those with short larval periods, species with low capture probabilities using dipnet or trapping techniques, or those with very limited or unpredictable breeding windows (Dorcas et al. 2009). Thus, anuran vocalizations collected via ARUs can further our understanding of how amphibians use created wetlands on restored mined lands.

The goal of this study was to assess the response of calling anurans to wetland creation and forest restoration on legacy surface mines in Kentucky and West Virginia (USA). The first objective was to examine mean amphibian occupancy, species-level occupancy, and species richness in created wetlands in young FRA [1-6 years post forest restoration] and older FRA [10-19 years post forest restoration] sites and compare to wetlands within unmined, mature forest sites. In addition, we also evaluated the effects of wetland size on mean occupancy, species richness, and species-level occupancy. Our second objective was to examine mean and species-specific detection probabilities, and factors influencing detection of calling anurans. In general, this study provides guidance on the effectiveness of wetland creation and reforestation of legacy surface mines for anurans, and information relevant for anuran monitoring in the Central Appalachian region.

6.2. Methods

6.2.1. Kentucky

We monitored frog calling activity at 19 wetlands sites on legacy mined lands and wetlands within mature, unmined forests in southeastern Kentucky (Figure 1). All wetlands were ≥ 200 m apart, and anuran choruses outside of the focal wetland were not counted during data processing (see below). We selected eight sites on Pine Mountain in Bell County (elevation 885 m). This land was part of a $\sim 40,500$ ha acquisition by The Nature Conservancy in 2019 called the “Cumberland Forest Project.” The Pine Mountain sites were mined for coal beginning in 2004, with grassland reclamation completed by 2014. These sites were re-forested via the FRA for legacy mined lands in 2022. Specifically, non-native species were removed, soil was de-compacted using a deep ripping shank, and 22 species of trees were planted at the site, including shortleaf pine (*Pinus echinata*), white oak (*Quercus alba*), chestnut oak (*Q. prinus*), and northern red oak (*Q. rubra*) (Green Forests Work 2022). At least 50 shallow wetlands were constructed at the site by first using small bulldozers to create “weep berms” which allow for water to pool in small areas on the upslope and then using excavators to dig depressions which intercept and retain precipitation and groundwater, as well as trap sediment (Green Forests Work 2022). These sites represent the young FRA (herein, YFRA-KY) sites in Kentucky.

We selected six sites on the Paul Van Booven Wildlife Management Area (WMA) in Breathitt County to represent the older FRA (herein, OFRA-KY) sites. The WMA is over ~ 800 ha with a maximum elevation of 470 m. The WMA was previously surfaced mined for coal from the late 1990s to the early 2000s. The FRA was applied to our six sites between 2003-2007. Sites were planted with white oak (*Q. alba*), white pine (*P. strobus*), white ash (*Fraxinus americana*), green ash (*F. pennsylvanica*), black locust (*Robinia pseudoacacia*), yellow poplar (*Liriodendron tulipifera*), and redbud (*Cercis canadensis*) (Agouridis et al. 2017). Four of the wetlands in this treatment were created to assist with hydrology (e.g. under drain) in a valley fill (Agouridis et al. 2017); 2 wetlands were created (via FRA methods explained above) during a stream restoration project.

We selected five wetlands in the London Ranger District of the Daniel Boone National Forest (DBNF; Jackson County, KY) as our unmined sites. The London Ranger District incorporates over $\sim 81,000$ ha in eastern Kentucky. Tree assemblage across the DBNF has been described as mesophytic mixed hardwoods forest type (Braun 1950). We used naturally-formed ridgetop wetlands for this treatment (Skribbe et al. 2021). These sites represent reference (herein, MAT-KY) category as they contained mature forests not mined for coal.

6.2.2. West Virginia

In West Virginia, we monitored calling anurans at eight study sites on legacy mined lands, and four sites within mature, unmined forests (Figure 1); again, all wetlands were ≥ 200 m apart and anuran chorus detected outside of the focal wetland were not counted in data processing. All sites were located on Cheat Mountain in the Monongahela National Forest (MNF; Randolph and Pocahontas Counties). Northern hardwood species, including American beech (*Fagus grandifolia*), sugar maple (*Acer saccharum*), and black cherry (*Prunus serotina*) are common in the area, with red spruce (*Picea rubens*) found along high peaks and ridgelines (Rentch et al. 2007; Stephenson and Clovis 1983). Legacy mined lands occurred on the Mower Tract (elevation 1478 m), which was intensively logged in the late 1800’s, surface mined for coal in the late 1970s and early 1980s, and reclaimed as grassland post-coal mining. Beginning in 2009, the FRA (see description above) was applied to the Mower Tract, and included non-native species removal, soil decompaction, coarse woody debris loading, planting with a seedling mix of red spruce and 12 hardwood species, and creation of over 800 wetlands. The West Virginia YFRA sites (i.e., YFRA-WV) were reforested between 2017-2020 ($n = 4$) and the OFRA sites (OFRA-WV) were reforested between 2013-2014 ($n = 4$). Mature forest sites (MAT-WV; $n = 4$) were located at similar elevations to the YFRA-WV and OFRA-WV sites within the MNF. See Lambert et al. (2021) and Sherman et al. (2024) for more details on the West Virginia study sites.

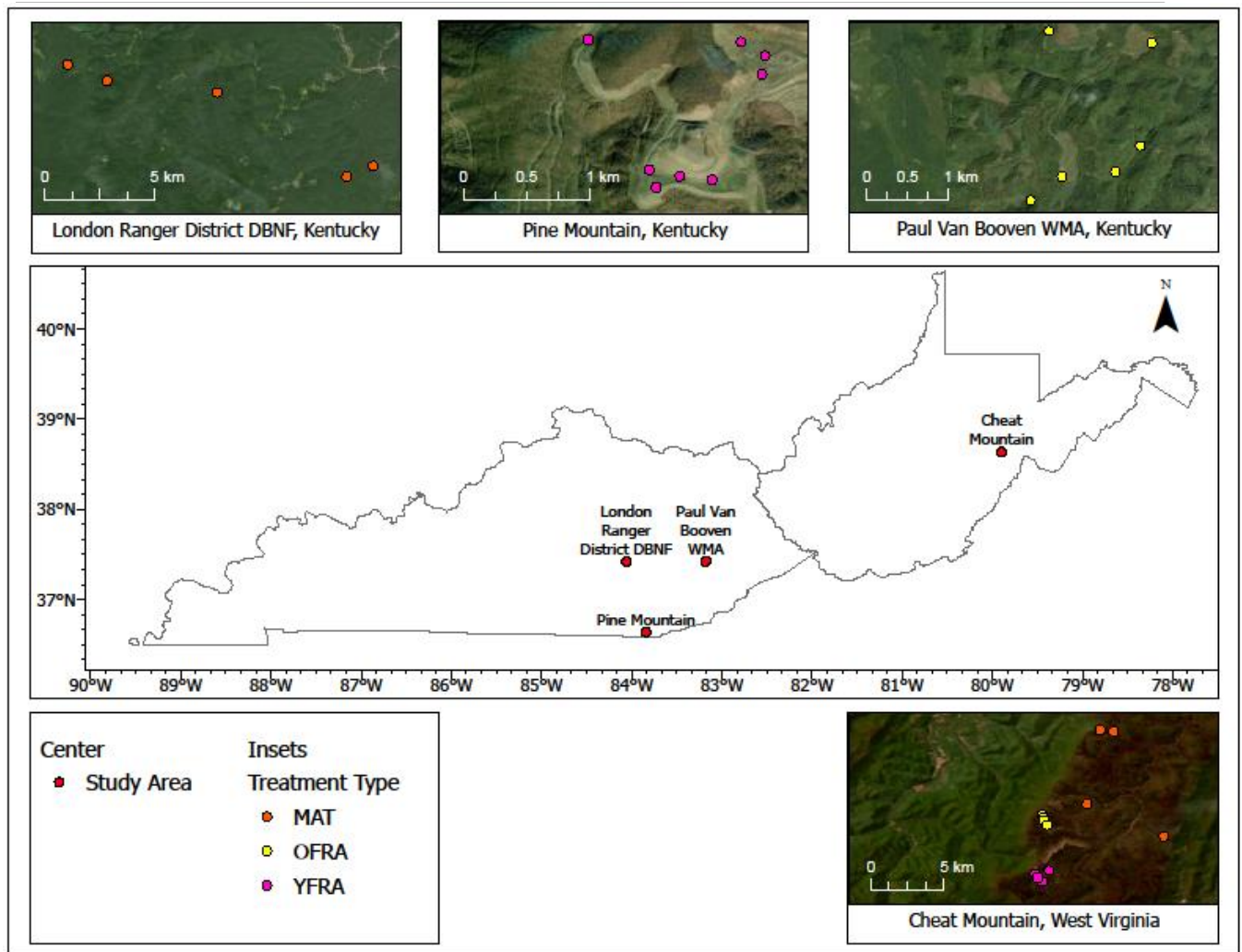


Fig 6. 1. Map of 19 sites across eastern Kentucky and 12 sites in eastern West Virginia, USA, each belonging to one of three treatment types. White sites represent young FRA sites that were reclaimed between 1 year ago (YFRA). Yellow sites represent old FRA sites that were reclaimed between 7-23 years ago (OFRA). Green sites represent unmined, mature forests (MAT).

6.2.3. Wetland Attributes

In June 2023, we collected several physical measurements at each wetland to examine how wetland dimensions, water chemistry, and canopy cover varied across treatment types. We measured wetland length and width at the widest points of the basin, and surface area (length \times width). Canopy cover was estimated in the middle of June 2022 using a spherical crown densiometer (Forestry Suppliers, Jackson, MS, USA), calculated as the average of four measurements taken facing each cardinal direction while standing in the approximate center of the wetland. We collected a 100 mL water sample from each wetland on three occasions (~once per week) during the field season. Water samples were kept on ice in the field, and frozen until they were transported to the University of Kentucky Department of Forestry and Natural Resources Hydrology lab. All water samples were analyzed for turbidity (FTU), conductivity ($\mu\text{S cm}^{-1}$), pH (H^+), Total Organic Carbon (TOC (mg/L)), Cl (mg/L), SO_4 (mg/L), $\text{NO}_3\text{-N}$ (mg/L), $\text{NH}_4\text{-N}$ (mg/L), Ca^{+2} (mg/L), Mg^{+2} (mg/L), K^+ (mg/L), Na^+ (mg/L), Mn (mg/L), Fe (mg/L), Al (mg/L), and $\text{NO}_2\text{-N}$ (mg/L). Water quality sampling, preservation, and analytic protocols were performed in accordance with standard methods (Greenberg et al. 1992).

6.2.4. Acoustic Surveys and Data Processing

In early January 2023, we placed one acoustic recorder (Song Meter Micro; Wildlife Acoustics) at each of the sites in Kentucky. Recorders were placed at West Virginia sites in early March, prior to snow melt. We

attached recorders 2 m off the ground to trees or garden stakes (with a wooden attachment) approximately 2 m from wetland edge. We equipped each recorder with a microSD card (SanDisc, 128GB), three batteries, and one dessicant pack for moisture control. We set ARUs to record 15 minutes each hour between 15:00 – 24:00. We checked recorders, microSD cards, and batteries once per month for malfunctions. Recorders were pulled from sites in mid-July 2023.

We randomly selected 15 recordings within each anuran breeding window (winter, spring, and summer) per site, for a total 15 recordings for each calling window ($n = 45$ total recordings for each site). This allowed us to partition the audio recordings into manageable number of samples for data processing and occupancy analyses. Before randomly selecting recordings, we eliminated dates when the daily high temperature was below 5.6°C and dates with wind speeds above 13 mph as these environmental conditions are either not suitable for anuran breeding activity, or produce inaudible recordings (Foreman et al. 2017). Calling windows varied by region due to differences in climate. In Kentucky, we considered the winter calling window from January 16 – March 30, the spring breeding window from April 1 – May 31, and the summer breeding window from June 1 – July 17. In West Virginia, we considered winter calling window to range from March 13-April 15, spring from April 16-May 31, and summer from June 1 through July 11. Within each calling window, we listened to different “hourly windows” to maximize species detection. In the winter calling window, we listened from 15:00-20:00, and in spring and summer windows we listened from 20:00-24:00. We processed recordings by listening to 5 minute interval in each randomly selected date and time for each site, and recorded all anuran species heard calling at the focal wetlands. Anuran choruses that were heard, but not at the focal wetland, were not included in the analysis. In total, we listened to a total of 855 recordings and 4275 listening minutes at the 19 Kentucky sites, and 540 recordings and 2700 listening minutes at the 12 West Virginia sites.

We collected variables known to contribute to the detection of anurans including day of year, time, and temperature (Lehtinen and Witter 2012; Dostine et al. 2013; Strain et al. 2017; Table 1). Day of year and time were recorded for every recording. Daily low temperatures were determined using Kentucky Mesonet (Mahmood et al. 2019). We used Jackson County for MAT sites, Breathitt County for OFRA sites, and Letcher County YFRA sites within the Kentucky Mesonet website. In West Virginia, air temperature data were collection from the Showshoe mesonet station (elevation 1428 m).

6.2.5. Data Analyses

6.2.5.1. Wetland Attributes

We performed an analysis of variance (ANOVA) in R (version 4.2.2; (R Core Team 2024) to examine differences among wetland attributes (i.e., wetland area, canopy cover, water chemistry) among our 4 treatment types (i.e., YFRA, OFRA, REGEN, MAT). Significance was assessed using an alpha level of 0.05. We performed a Tukey’s Honestly Significant Difference (HSD) test to further analyze significant differences detected by the ANOVA

6.2.5.2. Multi-species Occupancy Models

We used presence-absence data from recordings to develop a hierarchical multi-species community occupancy model to assess site covariates (treatment type and wetland area) and detection covariates (day of year, time of calling, and daily minimum temperature) on anuran communities (Zipkin et al. 2009; MacKenzie et al. 2017). The model output provides species-specific estimates of occupancy and detection as well as site-specific estimates of species richness (Dorazio and Royle 2005). Due to the shared breeding preferences of the anuran in this study, we used this approach as it borrows information from the more “data-rich” species and improves estimates for the more “data-poor” species (Pacifi et al 2014). However, we used separate models for each of study areas due to differences in the anuran communities known to occur in southeastern Kentucky and eastern West Virginia, respectively.

We formatted our data into a matrix(i,j,k), where species i was detected at site j on sampling occasion k . The occupancy probability (i.e., Ψ_{ij}) represented the probability of species i occurring at site j , modeled as a function of covariate parameters on α . p_{ijk} represented the probability of detecting species i at site j on sampling occasion k , modeled as a function of covariate parameters on β . To model multi-species occupancy, we used the following equation:

$$\text{logit}(Y_{ij}) = u_i + \alpha_{1i}YFRA_j + \alpha_{2i}OFRA_j + \alpha_{3i}Area_j$$

To model detection probability, we used the following equation:

$$\text{logit}(p_{ijk}) = v_i + \beta_{1i}DOY_{jk} + \beta_{2i}Time_{jk} + \beta_{3i}Temp_{jk}$$

The model contained 6 parameters: α_{1i} , α_{2i} , α_{3i} , β_{1i} , β_{2i} , and β_{3i} . Parameters α_{1i} , α_{2i} were effect of site types (YFRA and OFRA respectively), with “MAT” serving as the reference category. Parameter α_{3i} , was the effect of the continuous covariate Area (i.e., wetland area), standardized to have a mean of zero. The parameter β_{1i} was the effect of day of year, β_{2i} was the effect of time of day, and β_{3i} was the effect of minimum daily temperature on anuran detection probability. Detection covariates were also standardized to have a mean of zero.

We used Markov chain Monte Carlo sampling within Bayesian framework for our occupancy models (Link et al 2022). Priors were uninformative and uniformly distributed with a mean of -3 and a precision of 3. Three parallel chains were run, and we used the Gelman-Rubén statistic as a measure of convergence for the model (Gelman and Rubin 1992). Models were below 1.02, indicating model convergence (Gelman and Rubin 1992). Each chain was run for 100,000 iterations with a burn-in at 10,000 samples and thinned every 3 samples. The model generated 90,000 samples from the posterior distribution including the mean, standard deviation, and 95% credible interval (CIs). Covariates were considered biologically meaningful if the credible intervals of the parameter estimates did not overlap zero. The models were executed in Program R version 4.2.2 (R Core Team 2024) with package R2WinBUGS (Sturtz et al. 2005), which exported the data into WinBUGS version 1.4 (Spiegelhalter et al. 2002).

6.3. Results

6.3.1. Wetland Attributes

We found water quality in all wetlands examined was generally considered good, but highly variable among samples as indicated by large standard deviations (Tables 6.1 and 6.2). Nutrient concentrations in both areas restored using FRA techniques (YFRA and OFRA) were slightly elevated compared to those found in undisturbed natural wetlands (Table 6.1). Parameters of concern from coal mining operations (sulfate, Mn, Fe, and Al) were also generally low on all FRA sites but slightly elevated compared to the undisturbed natural wetland sites (Table 6.2). SC and pH were slightly elevated in the YFRA and OFRA sites over natural wetland sites.

Table 6.1. Mean (standard deviation) of wetland water quality nutrient analysis (n=19) collected during the summer of 2023. All units are in mg/L. BDL = below detection.

	Cl	Mg	Ca	K	Na	NO3-N	NH4-N	TOC
<i>YFRA</i>	0.9 (0.5)	6.8 (1.4)	18.7 (6.8)	5.8 (4.3)	0.8 (0.5)	BDL	0.1(0.1)	18.8(2.7)
<i>OFRA</i>	0.5 (0.3)	6.2 (1.9)	15.4 (8.3)	4.8 (3.7)	1.7(1.3)	0.02 (0.05)	0.6 (0.7)	14.5 (6.8)
<i>MAT</i>	1.8 (3.4)	0.8 (0.3)	2.7 (1.4)	2.2 (1.4)	2.0 (2.4)	0.01 (0.04)	1.5 (3.1)	10.8 (6.3)

Table 6.2. Mean (standard deviation) of wetland water quality analysis (n=19) collected during the summer of 2023. BDL = below detection.

	SO4	Mn	Fe	Al	pH	EC
	-----mg/L-----					μS/cm
<i>YFRA</i>	112.3 (79.6)	0.2 (0.5)	0.2 (1.8)	0.2 (0.4)	7.8 (0.2)	282.6 (152.6)
<i>OFRA</i>	91.3 (72.9)	0.3(1.2)	0.6 (1.0)	0.1 (0.5)	7.1 (0.8)	290.7 (240.5)
<i>MAT</i>	13.2 (10.7)	0.1 (0.07)	0.5 (0.3)	0.3 (0.5)	6.5 (0.3)	46.8(28.4)

6.3.2. Occupancy and Species Richness

We detected 10 anuran species in Kentucky using the acoustic recorders, with nine species detected at YFRA-KY sites, seven species at OFRA-KY sites, and six species at MAT-KY sites. Five anuran species were detected in West Virginia, and all five species were detected in each site type. Naïve occupancy, defined as the proportion of sampled sites where a species was detected on at least once occasion (Waddle et al. 2013), varied from 0.00 to 1.00 across all study sites (Table 6.3).

Table 6.3. Naïve Occupancy estimates for calling anurans in created wetlands on reforested surface mines in Kentucky and West Virginia (USA). Anurans were sampling using automated recording units across three treatment in each region. In Kentucky, YFRA sites were reclaimed in 2022, OFRA sites were reclaimed between 2000 and 2014, and MAT sites were unmined, mature forests. In West Virginia, YFRA sites were reclaimed between 2017-2020, OFRA sites were reclaimed between 2012 and 2015, and MAT sites were unmined, mature forests.

Scientific Name	YFRA-KY	OFRA-KY	MAT-KY	YFRA-WV	OFRA-WV	MAT-WV
<i>Anaxyrus americanus</i>	1.00	0.83	0.20	1.00	1.00	0.50
<i>Anaxyrus fowleri</i>	1.00	0.00	0.00	NA	NA	NA
<i>Scaphiopus holbrookii</i>	0.00	0.00	0.60	0.00	0.00	0.00
<i>Gastrophryne carolinensis</i>	0.38	0.00	0.00	NA	NA	NA
<i>Hyla chrysoscelis</i>	1.00	1.00	1.00	NA	NA	NA
<i>Hyla versicolor</i>	NA	NA	NA	0.75	0.50	0.25
<i>Pseudacris brachyphona</i>	1.00	0.83	0.00	NA	NA	NA
<i>Pseudacris crucifer</i>	1.00	1.00	1.00	1.00	1.00	1.00
<i>Lithobates clamitans</i>	1.00	0.33	1.00	1.00	1.00	0.75
<i>Lithobates palustris</i>	0.25	0.33	0.00	0.00	0.00	0.00
<i>Lithobates sylvaticus</i>	0.25	0.17	0.80	1.00	0.75	0.50

6.3.2.1. Kentucky

Mean anuran occupancy was $\Psi_{mean} = 0.54$ (CI = 0.21, 0.86) at our Kentucky study sites. The mean anuran occupancy varied across treatment types with $\Psi_{YFRA} = 0.88$ (CI = 0.65, 0.94), $\Psi_{OFRA} = 0.50$ (CI = 0.20, 0.82), and $\Psi_{MAT} = 0.55$ (CI = 0.21, 0.86)). The posterior distribution for YFRA-KY (1.98 (CI = 0.57, 2.86)) had positive influence on mean occupancy of anuran species (Figure 6.2). Overall, estimated mean species richness was greater in YFRA-KY site type than OFRA-KY or MAT-KY site types (Figure 6.3).

We found some effects of site type and wetland area on species-specific occupancy (Table 6.4). For example, *Anaxyrus americanus* (American Toad (2.23 (CI = 0.94, 2.95))), *A. fowleri* (Fowler's Toad (2.39 (CI = 1.35, 2.97))), *Gastrophryne carolinensis* (Eastern Narrow-mouthed Toad (2.12 (CI = 0.57, 2.93))), *Pseudacris brachyphona* (Mountain Chorus Frog (2.27 (CI = 1.04, 2.95))), and *Lithobates clamitans* (Green Frog (2.13 (CI = 0.58, 2.94))) occupancy was all positively associated with YFRA-KY. *Lithobates palustris* (Pickerel Frog (-2.67 (CI = -4.33, -0.98))) and *G. carolinensis* (-2.66 (CI = -4.39, -0.87)) occupancy was negatively associated with MAT-KY treatment, whereas *H. chrysoscelis* (Cope's Gray Treefrog (3.15 (CI = 1.13, 4.87))), *P. crucifer* (Spring Peeper (3.48 (CI = 1.43, 4.92))), and *L. clamitans* (Green Frog (1.73 (CI = 0.12, 3.52))) occupancy was positively associated with MAT-KY treatment. *Lithobates clamitans* (1.53 (CI = 0.06, 2.87)), *L. palustris* (1.58 (CI = 0.38, 2.80)), and *L. sylvaticus* (1.90 (CI = 0.48, 2.94)) were more likely to occupy larger wetlands.

Occupancy estimates varied among species (Table 6.5). For example, *H. chrysoscelis* had high

occupancy estimates at all treatment types ($\Psi_{\text{YFRA}} = 0.98$ (CI = 0.94, 0.99), $\Psi_{\text{OFRA}} = 0.96$ (CI = 0.80, 0.99), and $\Psi_{\text{MAT}} = 0.93$ (CI = 0.65, 0.99)). Conversely, the occupancy of *A. fowleri* varied among treatments ($\Psi_{\text{YFRA}} = 0.78$ (CI = 0.52, 0.94), $\Psi_{\text{OFRA}} = 0.13$ (CI = 0.02, 0.40), and $\Psi_{\text{MAT}} = 0.35$ (CI = 0.05, 0.79)).

Mean anuran detection was $p_{\text{mean}} = 0.144$ (CI = 0.05, 0.31), and was not strongly influenced by date (-0.03 (CI = -0.27, 1.98)), but was positively associated with time of day (0.23 (CI = 0.11, 0.37) and daily minimum temperature (0.19 (CI = 0.004, 0.343)) (Figure 6.2A). Baseline detection probabilities varied considerably among species, with *Scaphiopus holbrookii* (Eastern Spadefoot) having a low baseline detection of $p = 0.01$ (CI = 0.0002, 0.04) and *P. crucifer* having a baseline detection of $p = 0.59$ (CI = 0.55, 0.62)) (Table 6.6). Time of day had a significant effect on the detection of eight of the ten species of anurans. *Anaxyrus americanus*, *S. holbrookii*, *H. chrysoseleis*, *P. brachyphona*, *P. crucifer*, *L. clamitans*, *L. palustris*, and *L. sylvaticus* tended to call later in the evening (Table 6.6). Temperature had a significant effect on the detection of four of the ten species of anurans (Table 6.6). *Hyla chrysoseleis*, *P. brachyphona*, *P. crucifer*, and *L. clamitans* tended to call on evenings with warmer minimum temperatures (Table 6.6). Day of the year did not influence anuran species detection.

Table 6.4. Anuran species occupancy parameter estimates for site covariates (YFRA, OFRA, MAT, and wetland area) in Kentucky. 95% credible intervals (95% CI) that do not overlap zero are denoted by an asterisk. YFRA sites were reclaimed in 2022, OFRA sites were reclaimed between 2000 and 2014, and MAT sites were unmined, mature forests.

Scientific Name	YFRA (95% CI)	OFRA (95% CI)	MAT (95% CI)	Wetland Area (95% CI)
<i>Anaxyrus americanus</i>	2.23 (0.94, 2.95)*	0.98 (-0.94, 2.76)	0.18 (-1.18, 1.68)	-0.35 (-1.58, 0.81)
<i>Anaxyrus fowleri</i>	2.39 (1.35, 2.97)*	-1.23 (-2.87, 0.73)	-0.97 (-2.30, 0.36)	-0.16 (-1.32, 0.94)
<i>Scaphiopus holbrookii</i>	1.51 (-1.94, 2.90)	-0.55 (-2.79, 2.17)	-0.09 (-3.12, 3.86)	0.56 (-1.70, 2.65)
<i>Gastrophryne carolinensis</i>	2.12 (0.57, 2.93)*	-0.53 (-2.73, 1.81)	-2.66 (-4.39, -0.87)*	1.10 (-0.18, 2.51)
<i>Hyla chrysoseleis</i>	1.98 (-0.07, 2.92)	0.59 (-1.75, 2.75)	3.15 (1.13, 4.87)*	0.53 (-1.36, 2.45)
<i>Pseudacris brachyphona</i>	2.27 (1.04, 2.95)*	1.08 (-0.81, 2.80)	-0.10 (-1.47, 1.36)	-0.74 (-2.01, 0.43)
<i>Pseudacris crucifer</i>	1.95 (-0.15, 2.92)	0.50 (-1.87, 2.73)	3.48 (1.43, 4.92)*	0.53 (-1.44, 2.50)
<i>Lithobates clamitans</i>	2.13 (0.58, 2.94)*	-1.35 (-2.89, 0.61)	1.73 (0.12, 3.52)*	1.53 (0.06, 2.87)*
<i>Lithobates palustris</i>	1.86 (-0.21, 2.90)	1.20 (-0.90, 2.86)	-2.67 (-4.33, -0.98)*	1.58 (0.38, 2.80)*
<i>Lithobates sylvaticus</i>	1.43 (-1.49, 2.87)	-0.47 (-2.61, 1.73)	-0.68 (-2.44, 1.35)	1.90 (0.48, 2.94)*

Table 6.5. Mean estimated occupancy (Ψ) and 95% credible intervals for 10 anuran species across three treatment types in Kentucky. Amphibians were sampled via acoustic recorders in recently created wetlands on sites restored via the Forestry Reclamation Approach. YFRA sites that were reclaimed in 2022, OFRA sites were reclaimed between 2000 and 2014, and MAT sites were unmined, mature forests.

Scientific Name	YFRA (95% CI)	OFRA (95% CI)	MAT (95% CI)
<i>Anaxyrus americanus</i>	0.90 (0.72, 0.98)	0.73 (0.39, 0.95)	0.57 (0.14, 0.93)
<i>Anaxyrus fowleri</i>	0.78 (0.52, 0.94)	0.13 (0.02, 0.40)	0.35 (0.05, 0.79)
<i>Scaphiopus holbrookii</i>	0.67 (0.05, 0.99)	0.38 (0.01, 0.98)	0.50 (0.03, 0.98)
<i>Gastrophryne carolinensis</i>	0.39 (0.10, 0.77)	0.07 (0.002, 0.31)	0.13 (0.007, 0.52)
<i>Hyla chrysoscelis</i>	0.99 (0.94, 0.99)	0.96 (0.79, 0.99)	0.93 (0.65, 0.99)
<i>Pseudacris brachyphona</i>	0.87 (0.67, 0.97)	0.70 (0.35, 0.94)	0.52 (0.11, 0.91)
<i>Pseudacris crucifer</i>	0.99 (0.95, 0.99)	0.96 (0.82, 0.99)	0.95 (0.72, 0.99)
<i>Lithobates clamitans</i>	0.97 (0.88, 0.99)	0.58 (0.22, 0.89)	0.82 (0.39, 0.98)
<i>Lithobates palustris</i>	0.34 (0.07, 0.67)	0.23 (0.03, 0.57)	0.12 (0.01, 0.59)
<i>Lithobates sylvaticus</i>	0.65 (0.29, 0.96)	0.28 (0.03, 0.72)	0.41 (0.05, 0.91)

Table 6.6. Baseline detection probability for Kentucky anuran species and parameter estimates for detection covariates (day of the year, time, and minimum daily temperature). 95% credible intervals (95% CI) that do not overlap zero are denoted by an asterisk. YFRA sites were reclaimed in 2022, OFRA sites were reclaimed between 2000 and 2014, and MAT sites were unmined, mature forests. Anuran calling data were collected via automated recording units deployed from January-July 2023. Recordings were collected each hour from 1600 – 2400. A subset of 45 recordings were processed for this analysis.

Scientific Name	Baseline detection (p)			
	(95% CI)	Date (95% CI)	Time (95% CI)	Temp (95% CI)
<i>Anaxyrus americanus</i>	0.11 (0.08, 0.14)	-0.25 (-0.53, 0.01)	0.26 (0.10, 0.46)*	0.12 (-0.13, 0.33)
<i>Anaxyrus fowleri</i>	0.19 (0.15, 0.22)	0.05 (-0.24, 0.36)	0.15 (-0.08, 0.32)	0.08 (-0.22, 0.31)
<i>Scaphiopus holbrookii</i>	0.01 (0.002, 0.04)	0.09 (-0.38, 0.70)	0.26 (0.01, 0.59)*	0.23 (-0.12, 0.63)
<i>Gastrophryne carolinensis</i>	0.03 (0.02, 0.08)	0.12 (-0.33, 0.73)	0.20 (-0.13, 0.45)	0.20 (-0.16, 0.56)
<i>Hyla chrysoscelis</i>	0.25 (0.23, 0.28)	0.03 (-0.13, 0.20)	0.16 (0.01, 0.29)*	0.24 (0.09, 0.41)*
<i>Pseudacris brachyphona</i>	0.17 (0.15, 0.21)	0.10 (-0.11, 0.33)	0.24 (0.07, 0.41)*	0.23 (0.04, 0.44)*
<i>Pseudacris crucifer</i>	0.59 (0.55, 0.62)	-0.09 (-0.24, 0.06)	0.29 (0.17, 0.43)*	0.32 (0.17, 0.48)*
<i>Lithobates clamitans</i>	0.20 (0.17, 0.24)	0.15 (-0.05, 0.37)	0.21 (0.06, 0.35)*	0.26 (0.08, 0.45)*
<i>Lithobates palustris</i>	0.07 (0.04, 0.12)	-0.31 (-0.82, 0.08)	0.31 (0.09, 0.68)*	0.07 (-0.41, 0.36)
<i>Lithobates sylvaticus</i>	0.04 (0.02, 0.07)	-0.24 (-0.70, 0.12)	0.26 (0.05, 0.54)*	0.16 (-0.18, 0.44)

6.3.2.2. West Virginia

Mean anuran occupancy was $\Psi_{mean} = 0.73$ (CI = 0.40, 0.94) at our study sites in West Virginia. The posterior distribution for YFRA-WV (1.74 (CI = -0.03, 2.83)) indicated a mostly positive relationship, suggesting the mean occupancy was greatest in the YFRA-WV site type (Figure 6.2B). The mean anuran occupancy for each treatment type with $\Psi_{YFRA} = 0.85$ (CI = 0.21, 0.94), $\Psi_{OFRA} = 0.81$ (CI = 0.39, 0.94), and $\Psi_{MAT} = 0.76$ (CI = 0.40, 0.95). Estimated mean species richness was greater in YFRA-WV treatment type than OFRA or MAT sites (Figure 6.3B).

We found some treatment effects on species-specific occupancy. *Lithobates sylvaticus* (1.40 (CI = 0.026, 2.916)) occupancy was positively associated with the YFRA-WV site type, whereas *L. clamitans* (1.55 (CI = 0.05, 3.47)) and *P. crucifer* (2.521 (0.51, 4.74)) occupancy was positively associated with the MAT-WV site type (Table 6.7). Wetland area did not have an effect on species-specific occupancy in West Virginia. Occupancy estimates were ≥ 0.70 for all species across site types (Table 6.8).

Mean anuran detection was $p_{mean} = 0.17$ (CI = 0.02, 0.59), and was not influenced by date (-0.01 (CI = -2.10, 2.10) or time of day (0.52 (CI = -0.95, 1.92) (Figure 6.2B). Mean anuran detection was positively associated with daily minimum temperature (0.71 (CI = 0.13, 1.41)) (Figure 6.2B). Baseline detection probabilities were generally quite low ($p \leq 0.13$) for most species, with the exception of *P. crucifer* with a baseline detection of $p = 0.57$ (CI = 0.54, 0.63)) (Table 6.9). Time of day had a positive effect on detection of *A. americanus*, *H. versicolor*, *P. crucifer* and *L. clamitans*; only the *L. sylvaticus* had a negative association with time of day (-1.05 (CI = -1.7, -0.43)) (Table 6.9). Temperature was positively associated with 4 of 5 species, and posterior distribution for temperature (0.309 (CI = -0.02, 0.62)) on *P. crucifer* detection indicated a mostly positive relationship (Table 6.9). Day of year had a negative relationship *L. sylvaticus* detection (-2.565 (CI = -2.99, -1.72)) and a positive association with *H. versicolor* detection (1.84 (CI = 0.41, 2.93)), suggesting these species vocalize earlier and later in the sampling period, respectively (Table 6.9).

Table 6.7. Anuran species occupancy parameter estimates for site covariates (YFRA, OFRA, MAT, and wetland area) in West Virginia. 95% credible intervals (95% CI) that do not overlap zero are denoted by an asterisk. YFRA sites were reclaimed in between 2017-2020, OFRA sites were reclaimed between 2012 and 2015, and MAT sites were unmined, mature forests.

Scientific Name	YFRA (95% CI)	OFRA (95% CI)	MAT (95% CI)	Wetland Area (95% CI)
<i>Anaxyrus americanus</i>	1.78 (-0.18, 2.908)	1.623(-0.280, 2.891)	1.022(-0.458, 2.627)	-0.640(-2.468, 1.107)
<i>Hyla versicolor</i>	1.759(-0.219, 2.901)	1.350(-0.980, 2.844)	0.031(-2.018, 2.573)	-0.251(-2.032, 1.704)
<i>Pseudacris crucifer</i>	1.704(-0.519, 2.902)	0.50 (-1.87, 2.73)	2.521 (0.508, 4.743)*	-0.370 (-2.312, 1.727)
<i>Lithobates clamitans</i>	1.760(-0.278, 2.905)	1.561(-0.435, 2.883)	1.547 (0.054, 3.466)*	-0.275 (-2.070, 1.712)
<i>Lithobates sylvaticus</i>	1.395(0.026, 2.916)*	-0.47 (-0.768, 2.849)	0.651 (-0.929, 2.504)	-0.125(-1.789, 1.825)

Table 6.8. Mean estimated occupancy (Ψ) and 95% credible intervals of 5 anuran species across three sites types in West Virginia. Amphibians were sampled via acoustic recorders in recently created wetlands on sites restored via the Forestry Reclamation Approach. YFRA sites were reclaimed in between 2017-2020, OFRA sites were reclaimed between 2012 and 2015, and MAT sites were unmined, mature forests.

Scientific Name	YFRA (95% CI)	OFRA (95% CI)	MAT (95% CI)
<i>Anaxyrus americanus</i>	0.92 (0.67, 0.99)	0.91 (0.67, 0.99)	0.85 (0.42, 0.99)
<i>Hyla versicolor</i>	0.80 (0.38, 0.99)	0.74 (0.27, 0.99)	0.70 (0.15, 0.99)
<i>Pseudacris crucifer</i>	0.89 (0.62, 0.99)	0.84 (0.50, 0.99)	0.80 (0.32, 0.99)
<i>Lithobates clamitans</i>	0.94 (0.75, 0.99)	0.93 (0.74, 0.99)	0.89 (0.55, 0.99)
<i>Lithobates sylvaticus</i>	0.89 (0.62, 0.99)	0.84 (0.50, 0.99)	0.80 (0.32, 0.99)

Table 6.9. Baseline detection probability for West Virginia anuran species and parameter estimates for detection covariates (day of the year, time, and minimum daily temperature). 95% credible intervals (95% CI) that do not overlap zero are denoted by an asterisk. YFRA sites were reclaimed between 2017-2020, OFRA sites were reclaimed between 2012-2015, and MAT sites were unmined, mature forests.

Scientific Name	Baseline detection (p) (95% CI)	Date (95% CI)	Time (95% CI)	Temperature (95% CI)
<i>Anaxyrus americanus</i>	0.10 (0.05, 0.13)	-0.314 (-0.955, 0.319)	1.301 (0.806, 1.837)*	0.803 (0.336, 1.331)*
<i>Hyla versicolor</i>	0.01 (0.001, 0.02)	1.841 (0.414, 2.926)*	0.737 (0.01, 1.557)*	0.857 (0.108, 1.871)*
<i>Pseudacris crucifer</i>	0.57 (0.54, 0.63)	-0.111 (-0.498, 0.276)	1.035 (0.785, 1.293)*	0.309 (-0.002, 0.617)
<i>Lithobates clamitans</i>	0.13 (0.09, 0.19)	1.096 (0.524, 1.698)	0.624 (0.30, 0.965)*	0.883 (0.43, 1.396)*
<i>Lithobates sylvaticus</i>	0.02 (0.01, 0.05)	-2.565 (-2.986, -1.718)*	-1.05 (-1.733, -0.428)*	0.707 (0.234, 1.218)*

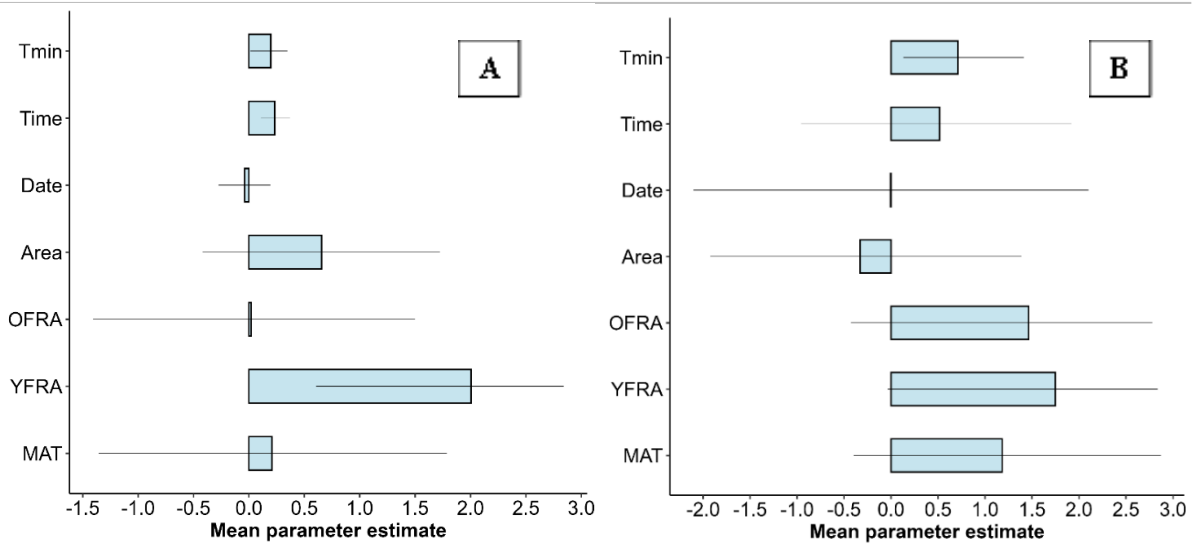


Figure 6.2. Mean parameter estimates for mean anuran occupancy in Kentucky (A) and West Virginia (B) wetlands. Blue bars to the right of zero represent a positive relationship between the species occupancy and the given parameter while bars to the left of zero indicate a negative relationship. Estimates are only considered meaningful if the credible intervals do not overlap zero. In Kentucky (A), YFRA sites were reclaimed in 2022, OFRA sites were reclaimed between 2000 and 2014, and MAT sites were unmined, mature forests. In West Virginia (B), YFRA sites were reclaimed in between 2017-2020, OFRA sites were reclaimed between 2012 and 2015, and MAT sites were unmined, mature forests.

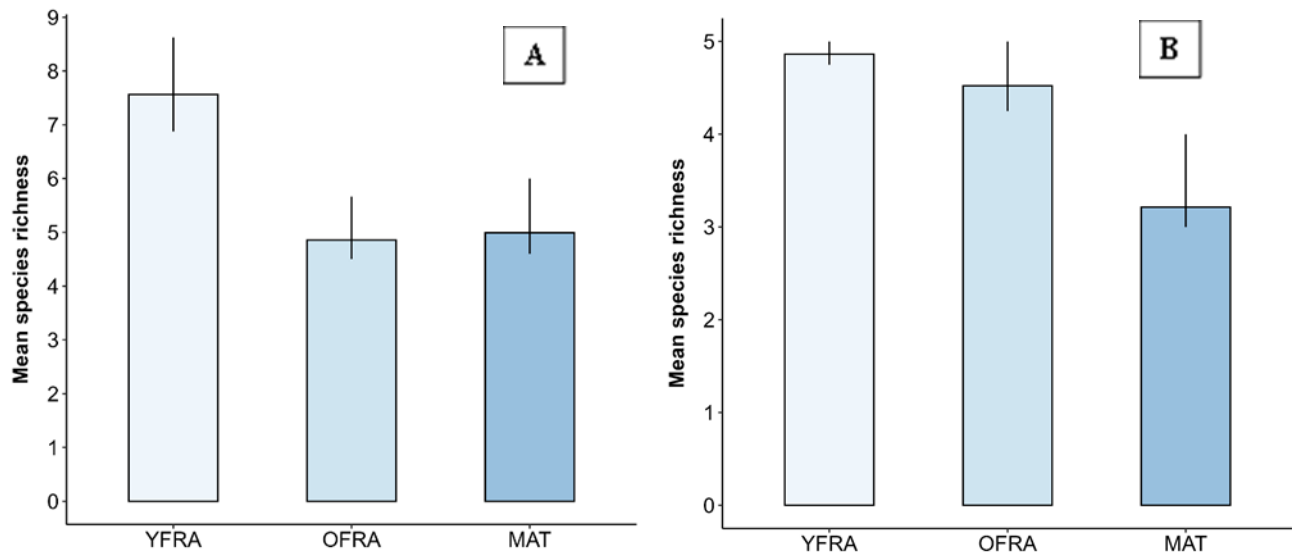


Figure 6.3. Mean calling anuran species richness across treatment types (KY A) and WV (B). In Kentucky (A), YFRA sites were reclaimed in 2022, OFRA sites were reclaimed between 2000 and 2014, and MAT sites were unmined, mature forests. In West Virginia (B), YFRA sites were reclaimed in between 2017-2020, OFRA sites were reclaimed between 2012 and 2015, and MAT sites were unmined, mature forests.

6.4. Discussion

Our results indicate that wetlands created on reforested surface mines provide habitat for most vocalizing anuran species found in the Central Appalachian region. Mean occupancy across all species was generally high (>0.50), and mean anuran occupancy and species richness were found to be greatest in the YFRA site type in both regions. Common species (i.e., *A. americanus*, *H. chrysoscelis/versicolor*, *P. crucifer*, *L. clamitans* and *L. sylvaticus*) that occurred in both Kentucky and West Virginia were detected vocalizing from wetlands across the site types (YFRA, OFRA, MAT). These common species generally exhibited mean occupancy probabilities that exceeded > 0.80 in wetlands, particularly in wetlands located within the YFRA site type. Species rarely detected via our surveys were not detected in some site types. For example, *S. holbrooki* and *G. carolinensis* were only detected at MAT-KY and YFRA-KY, respectively. However, due to low detection probabilities and the use of the multi-species occupancy model, rare species have some probability of occupancy because inferences about an individual species borrows strength from observations of other species within the community (Zipkin et al. 2009).

Previous studies have documented anurans using created wetlands for calling, with many showing similarity in species occupancy and richness between created and natural wetlands (See Brown et al. 2012). For example, in West Virginia (USA), human-created and beaver-created wetlands did not differ in occupancy of calling anuran occupancy (Strain et al. 2017), and species richness and abundance were higher in created mitigation wetlands compared to reference wetlands (Balcombe et al. 2005). Furthermore, Lehtinen and Galatowitsch (2001) found that amphibians rapidly colonized restored wetlands in an agricultural landscape, with 8 of 12 regional species establishing breeding populations within months of wetlands restoration. Although relatively few studies have compared pond-breeding amphibians on previously mined lands to unmined areas, Buckardt et al. (2023) demonstrated that amphibian community composition and species occupancy of wetlands is similar in mined and unmined landscapes. Our results suggest that calling anurans were able to rapidly colonize wetlands on previously mined land in the Central Appalachian region that underwent restoration 1-6 yrs prior to our surveys (i.e., YFRA). We attribute the rapid colonization to restoration techniques used at our study sites, which included reforestation, woody debris loading, and the creation of > 850 semi-permanent shallow wetlands. These restoration techniques likely enhanced terrestrial and aquatic habitat for anurans, and coupled with the fact that our restoration sites were in close proximity to national forest boundaries (i.e., MNF) or state natural areas (Pine Mt. in KY), likely facilitated colonization. However, we caution that multiple years of data may be needed to fully assess the suitability of created wetlands on mined lands for anurans as some species may rapidly colonize and breed in created wetlands only to disappear from these same sites over time (i.e., Oja et al. 2021).

We acknowledge that anuran calling activity does not always equate to successful reproduction and recruitment (i.e., Greenberg and Tanner 2005), which may be better indicators of restoration success. However, Sherman et al. (2024) conducted dipnet surveys for larval and adult amphibians at our West Virginia sites in 2022 and estimated the following mean occupancy probabilities: *A. americanus* $\Psi_{YFRA} = 0.12$, $\Psi_{OFRA} = 0.08$, and $\Psi_{MAT} = 0.06$; *L. clamitans* $\Psi_{YFRA} = 0.50$, $\Psi_{OFRA} = 0.61$, and $\Psi_{MAT} = 0.47$; *L. sylvaticus* $\Psi_{YFRA} = 0.42$, $\Psi_{OFRA} = 0.47$, and $\Psi_{MAT} = 0.37$; *P. crucifer* $\Psi_{YFRA} = 0.26$, $\Psi_{OFRA} = 0.28$, and $\Psi_{MAT} = 0.18$; *H. versicolor* $\Psi_{YFRA} = 0.44$, $\Psi_{OFRA} = 0.48$, and $\Psi_{MAT} = 0.24$ in created wetlands. In addition, Lambert et al. (2021) conducted larval sampling at some of the same sites in 2018, and detected larva of the 5 anuran species we detected via ARU recordings across various wetland ages. These previous studies (Lambert et al. 2021; Sherman et al. 2024) attributed larval amphibian occupancy in created wetlands to water chemistry within a normal range for the region (Sherman et al. 2024), restoration techniques that included planting of native herbaceous species around wetlands (Branduzzi et al. 2020), and the creation of seasonal wetlands, which are preferable breeding habitat for many amphibian species (Semlitsch 2002). Collectively, it appears that anurans are using created wetlands for both vocalizing and reproduction.

In Kentucky, our results suggested that sites with larger wetlands had a higher occupancy of *L. clamitans*, *L. palustris*, and *L. sylvaticus* compared to sites with smaller wetlands. Larger wetlands likely provide more cover (i.e., plants, leaf litter, etc.) for larva and space for territories for vocalizing adults (Martof 1953; Otto et al. 2007; Shulse 2010). Previous investigations have also highlighted the relationship between larger wetlands and increased occupancy rates and abundance (Lambert et al. 2021; Drayer et al. 2020; Semlitsch et al. 2015; Sherman et al. 2024). Specifically, seasonal wetlands that range from 1000–10,000 m² support the greatest number of species (Semlitsch et al. 2015), because the larger wetland size leads to

decreased inter- and intra-species competition (Millikin et al. 2019). The wetlands at our study site ranged from 6.6 m² to 252 m², suggesting that creating larger seasonal wetlands during restoration activities may promote higher anuran occupancy rates for some species.

Our low baseline detection probabilities for most species (except *P. crucifer*) was due to strong patterns in seasonal breeding activity for most species (Droege and Eagle 2005; de Solla et al. 2005; Dorcas et al. 2009). Although date was not a strong predictor of species-level detection in Kentucky, the majority of the distribution of parameter estimates for *A. americanus*, *L. sylvaticus*, *L. palustris*, and *P. crucifer* were negative, indicating preference for calling in winter or early spring. In West Virginia, we found that *L. sylvaticus* and *H. versicolor* were more detectable in winter/early spring and summer sampling periods, respectively. As noted in numerous previous studies (i.e., Oseen and Wassersug 2002; Steelman and Dorcas 2010; Feldman et al. 2023), we found that time of day and minimum daily air temperature were important detection covariates for many anuran species. Most species in both regions called later in the day, with the exception of *L. sylvaticus* in West Virginia. Furthermore, parameter estimates for minimum air temperature were positively associated with 4 of 10 anurans in Kentucky and all 5 species in West Virginia. Understanding the influence of date, time of day and minimum daily temperature allows for the precise targeting of certain species found in the Central Appalachians; for example, targeted efforts for *L. sylvaticus* in West Virginia should focus on warm afternoon or early evening hours in late winter or early spring.

6.4.1. Management Implications

In general, our results indicate that the FRA, coupled with the creation of seasonal wetlands restores habitat for calling anurans within a relatively short timeframe (1-6 yrs). The goal of FRA is to restore forests and ecosystem function on mined lands that were previously in a state of arrested succession. However, the creation of a high number of wetlands (>850), soil decompaction, woody debris loading, and planting of native tree species and grasses, the breeding habitat requirements of most anuran species in the region are also being provided. Ultimately, our results indicate that FRA and associated techniques may serve as an important amphibian conservation tool to promote biodiversity on legacy surface mines.

6.5. References

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Section 7. Publications and Products

7.1. Published Manuscripts

Sherman, L., C.D. Barton, J.C. Guzy, R.N. Davenport, J.J. Cox, J.L. Larkin, T. Fearer, J.C. Newman and S.J. Price. 2024. Wetland creation and reforestation of legacy surface mines in the Central Appalachian Region (USA): A potential climate-adaptation strategy for pond-breeding amphibians. *Water* 16:1202.

Snyder, B., C. Barton, M. Lacki, S.J. Price, and Z. Hackworth. 2024. Bat Activity on High elevation reforested coal mines in the Monongahela National Forest, West Virginia. *Ecological Restoration* 42:108-122.

Price, S.J., R.N. Davenport, L. Sherman, J.L. Larkin, J. Cox, J.C. Newman, and C. Barton. 2024. Response of Red-backed Salamanders (*Plethodon cinereus*) to the Forestry Reclamation Approach on legacy surface mines in the Monongahela National Forest (West Virginia). *Ecological Restoration* 42:105-107.

7.1.2. Manuscripts in Review

Davenport, R. N. , C.D. Barton, J.J. Cox, J. C. Guzy, L. Sherman, J. L. Larkin, T. Fearer, S.J. Price. *In review*. Effects of forest reclamation on avian guilds occupying legacy surface mines. *Restoration Ecology*.

7.1.3. Manuscripts in Preparation

Davenport, R. N. , C.D. Barton, J.J. Cox, J. C. Guzy, L. Sherman, J. L. Larkin, T. Fearer, S.J. Price. *In preparation*. Does forest reclamation provide habitat for Appalachian breeding birds on reclaimed surface mines? An evaluation of avian abundance in red spruce forest. Planned submission to *Ornithological Applications*.

Price, S.J., B. Snyder, L. Sherman, V. Burgess, J.L. Larkin, J. Cox, and C. Barton. *In preparation*. Mammals detected via automated recording units and camera traps on reforested surface mines in West Virginia and Kentucky (USA). Planned submission to *Reclamation Sciences*.

Newman J.C., Price, S.J., , J.C. Guzy, J.L. Larkin, J. Cox, , and C. Barton. *In preparation*. Restored wetlands on reforested surface mines provide breeding habitat for anurans in the Central Appalachians. Planned submission to *Journal of Wildlife Management*.

7.2 Presentations

7.2.1. Invited Presentations

Price, S.J. 2023. Coal Mining and Amphibians. Lessons from 10 years of research in the Central Appalachians. Eastern Kentucky University. October 27.

Price, S.J. 2024. Coal Mining and Amphibians: Lessons from 10+ years of research in the Central Appalachians. Department of Biology, Southern Illinois University. 7 October.

Price, S.J. 2024. Coal Mining and Amphibians: Lessons from 10+ years of research in the Central Appalachians. Department of Forestry, University of Kentucky. 2 October.

7.2.2. Poster/Talks

Varias, M., R.N. Davenport, C.D. Barton, J.J. Cox, L. Sherman, J. L. Larkin, T. Fearer, and S.J. Price. 2024. Effects of the Forestry Reclamation Approach on avian community composition in Appalachia. American Society of Reclamation Sciences 41st Annual Meeting. Knoxville TN. (Poster) June 1-5.

Snyder, B., C. Barton, M. Lacki, S.J. Price and Z. Hackworth. 2024. Bat activity on high elevation reforested coal mines in the Monongahela National Forest, West Virginia. American Society of Reclamation Sciences 41st Annual Meeting. Knoxville TN. (Talk) June 1-5.

Price, S.J. , L. Sherman, J. Newman, R. Davenport, J. Cox, J. L. Larkin, and C. Barton. 2024. Pond-breeding

amphibian response to the Forestry Reclamation Approach and wetland creation on legacy surface coal mines in the Central Appalachians. American Society of Reclamation Sciences 41st Annual Meeting. Knoxville TN. (Talk) June 1-5

Varias, M.J., R. N. Davenport, C.D. Barton, J.J. Cox, L. Sherman, J. L. Larkin, T. Fearer, S.J. Price. 2024. Avian Community Response to the Forestry Reclamation Approach on Legacy Surface Mines in the Central Appalachians. Central Hardwoods Conference. Lexington KY. (Talk) March 12-13.

Davenport, R. N., C.D. Barton, V. Burgess, J.J. Cox, T. Fearer, J. L. Larkin, L. Sherman, M. Varias, and S.J. Price. 2024. Effects of the Forestry Reclamation Approach on avian community composition in Appalachia. The Wildlife Society – Kentucky Chapter Annual Meeting. February 22-23.

Sherman, L., C.D. Barton, V. Burgess, J.J. Cox, T. Fearer, J. L. Larkin, R. N. Davenport, M. Varias, and S.J. Price. 2023. Pond-breeding amphibian responses to forest reclamation approach on reclaimed surface mines. The Wildlife Society Annual Meeting. (Poster) Louisville KY. November 5-9.

Davenport, R. N., C.D. Barton, V. Burgess, J.J. Cox, T. Fearer, J. L. Larkin, L. Sherman, M. Varias, and S.J. Price. 2023. Effects of the Forestry Reclamation Approach on avian community composition in Appalachia. The Wildlife Society Annual Meeting. (Poster) Louisville KY. November 5-9

Davenport, R. N., C.D. Barton, V. Burgess, J.J. Cox, T. Fearer, J. L. Larkin, L. Sherman, M. Varias, and S.J. Price. 2023. Effects of the Forestry Reclamation Approach on avian community composition in Appalachia. American Ornithological Society & Society of Canadian Ornithologists Joint Conference. (Poster) London, Ontario, Canada. August 8-12.

Davenport, R. N., C.D. Barton, V. Burgess, J.J. Cox, T. Fearer, J. L. Larkin, L. Sherman, M. Varias, and S.J. Price. 2023. Effects of forest reclamation on avian breeding habitat assemblages within reclaimed surface mines. (Poster) Wilson Ornithological Society 2023 Annual Meeting. Allentown, PA. June 20-23.

Davenport, R. N., C.D. Barton, V. Burgess, J.J. Cox, T. Fearer, J. L. Larkin, L. Sherman, M. Varias, and S.J. Price. 2023. Effects of forest reclamation and landscape features on avian occupancy, species richness, and abundance in Appalachia. Association of Southeastern Biologists 2023. (Poster) Winston-Salem, NC. March 23-26.

Davenport, R. N., C.D. Barton, V. Burgess, J.J. Cox, T. Fearer, J. L. Larkin, L. Sherman, M. Varias, and S.J. Price. 2023. Effects of forest reclamation and landscape features on avian occupancy, species richness, and abundance in Appalachia. Kentucky Chapter of The Wildlife Society Annual Meeting. (Poster) Barren River State Park, KY. February 23-24.

Sherman, L., C.D. Barton, V. Burgess, J.J. Cox, T. Fearer, J. L. Larkin, R. N. Davenport, M. Varias, and S.J. Price. 2023. Pond-breeding amphibian responses to forest reclamation approach on reclaimed surface mines. Southeast Partners in Amphibian and Reptile Conservation. (Poster) Black Mountain, NC. February 23-26.

7.3. Popular Articles/Media/Site Visits

Kentucky Field Notes Spring/Summer 2024. Restoring Wetlands in the Mountains. The Nature Conservancy. <https://www.nature.org/content/dam/tnc/nature/en/documents/2024-Spring-Field-Notes-TNC-KY.pdf>

Appalachian Mountains Joint Venture. Partnership Year in Review. 2023. https://amjv.org/wp-content/uploads/2024/07/FINAL-2023-AMJV-Partnership-Year-in-Review_June302024_.pdf

Appalachian Regional Reforestation Initiative Annual Conference held at the Monongahela National Forest on August 30, 2023. Over 50 in attendance. Led and organized day-long field tour for all participants.

National Fish and Wildlife Foundation and US Fish and Wildlife Field Review at the Monongahela National Forest on August 13, 2024. Over 20 in attendance. Led and organized day-long field tour for all participants.

7.4. Student Awards

Maddie Varias.

Best Student Presentation. American Society of Reclamation Sciences 41st Annual Meeting. Knoxville TN. June 1-5.

Rebecca Davenport

2023: American Ornithological Society Research Award, \$2500

2023: Karri Casner Environmental Sciences Fellowship, \$4773

2023: Kentucky Chapter of The Wildlife Society Student Development Award, \$300

2023: American Ornithological Society Travel Award, \$693

2023: Wilson Ornithological Society Travel Award, \$500

2023: Graduate Student Conference Travel Award, \$603.80

2023: Graduate Student Conference Travel Award, \$789

2023: Graduate Student Congress Conference Award, \$400

2023: Graduate Student Conference Travel Award, \$202.50

2022: Richards Graduate Student Research Activity Award, \$1500
