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RESEARCH ARTICLE

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ABSTRACT

The Forestry Reclamation Approach (FRA) is a practical guide to reforesting surface mined lands. Bats, a group of mammals with declining populations, could benefit from mine reforestation. To determine if the FRA can provide suitable bat foraging habitat, we surveyed bat activity at created depressional wetlands on 1-year-old and 8-year-old FRA reforested lands (FRA1; FRA8), wetlands in naturally regenerating forest on traditionally reclaimed mined land (~40 years old; REGEN), and wetlands in mature forest not previously mined (MAT). We passively recorded echolocation calls for 12 nights across 16 sites between June and August 2021. We analyzed bat activity using the number of recordings, pulses, and feeding buzzes in conjunction with nocturnal insect abundance and biomass, microhabitat characteristics, and landscape characteristics via generalized linear mixed effects modeling. Both FRA1 and FRA8 had activity levels similar to MAT. REGEN had significantly greater foraging activity than the other three land classes, possibly due to its distance from roads and proximity to forest edges. Insect abundance and biomass were comparable across sites, indicating FRA practices do not hinder the establishment of a prey base for bats. Overall, bats are utilizing the restored mined land for foraging. Reforestation of mined lands, complemented with wetland creation, provides habitat that could benefit bat species conservation in Appalachia.

Keywords: acoustic monitoring, created wetlands, legacy mined land, reforestation

Restoration Recap

- Wetlands created on reforested legacy mines were ponded throughout the growing season, exhibited good water quality, and were utilized by bats for foraging. Detection of bats and insects at the created wetlands was similar to levels observed in adjacent natural wetlands.
- Naturally formed wetlands on narrow mine benches undergoing natural succession exhibited the highest bat activity levels likely due to the presence of habitat conditions preferred by bats.
- Reclaimed coal mines in Appalachia can be restored or altered to attract wildlife species of concern through the intentional planting of native species and creation of habitat features such as wetlands.

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 (Shartitz et al. 2014). Though wetlands are an

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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 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 rouse 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 (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.

Methods

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 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 1970s 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

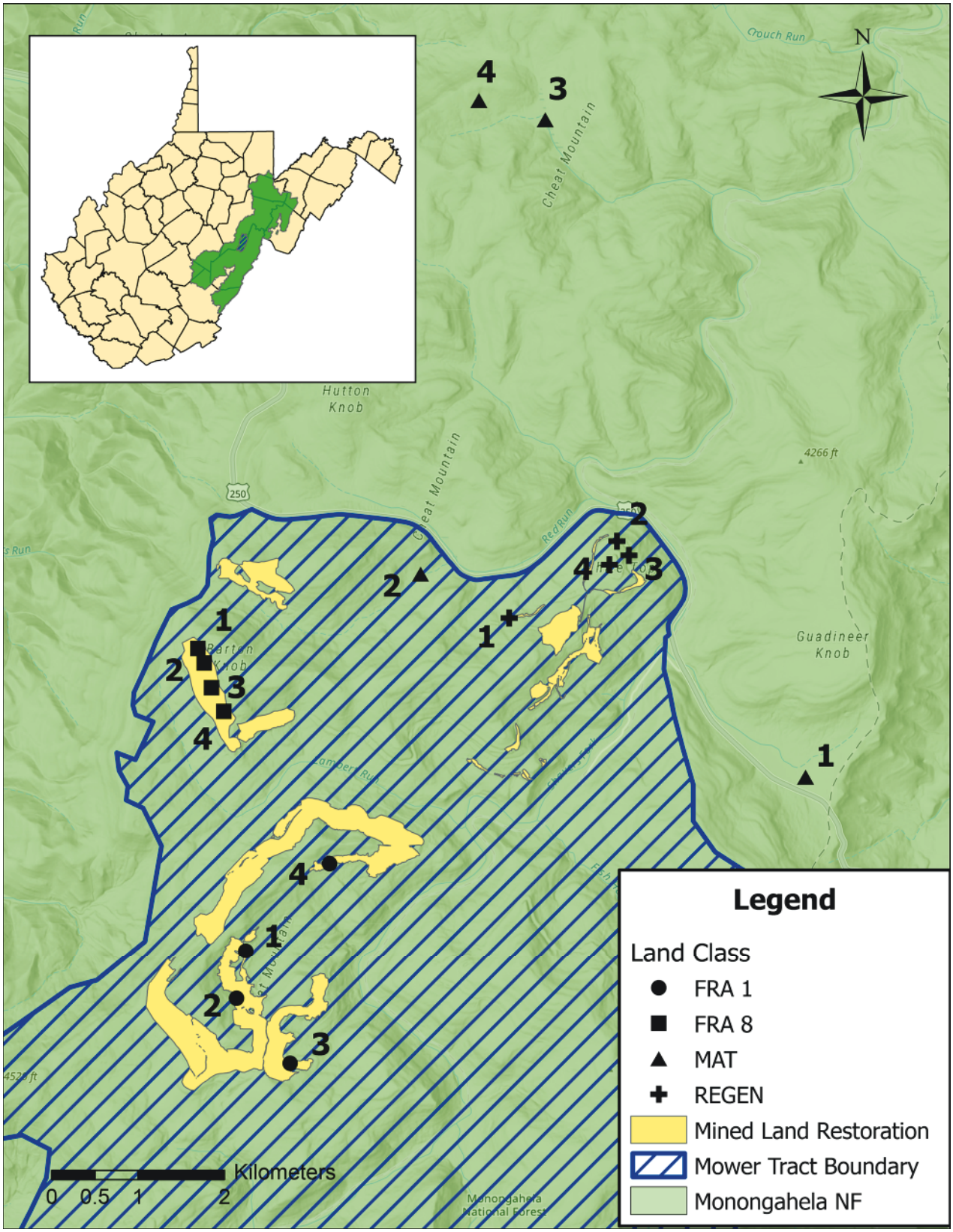

Figure 1. Map of the study area. The inset map denotes the Monongahela National Forest in east-central West Virginia. The hatched region is the Mower Tract, a parcel of land that was surface mined for coal in the 1970s. The main map shows the survey sites in relation to the Mower Tract. The solid filled regions have been restored using the FRA over a 12-year period. This study examined four land classes: 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).

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 m^2 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.

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 (see Supplementary Material, Figures S1–S4 for an examples of wetlands from each land class). 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.

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.

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.

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 University of Kentucky's Department of Forestry Hydrology Laboratory. Water quality testing included specific conductivity (SC µS/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).

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

Table 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

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 (ΔAIC_c), and weights of models (AIC_w). Functions in the AICcmodavg package (Mazerolle 2020) were used to calculate these metrics. We considered models with ΔAIC_c differences ≤ 6 as minimally supported and ΔAIC_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.

Results

Bat Acoustic Survey

Each site was 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 and Figure 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 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). 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.

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

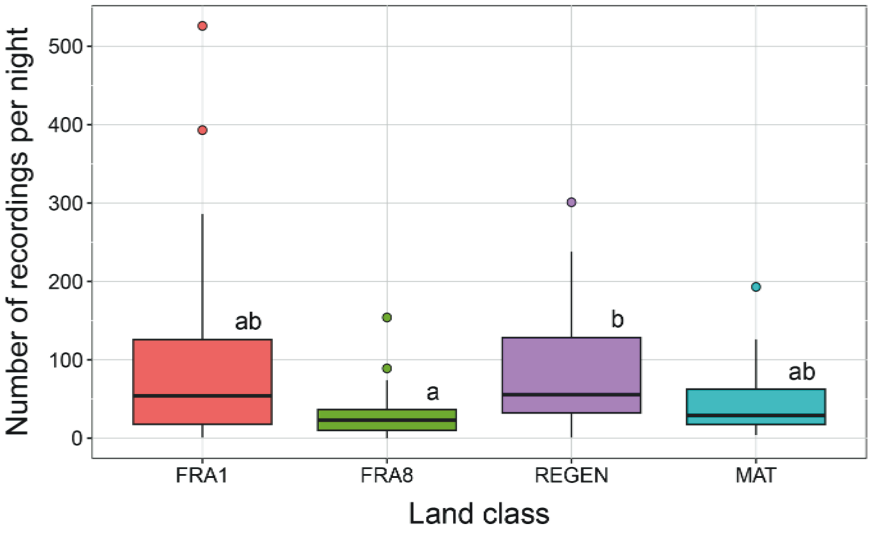

Figure 2. Boxplots showing the distribution of nightly recordings of each land class. Letters denote significance difference ($p < 0.05$).

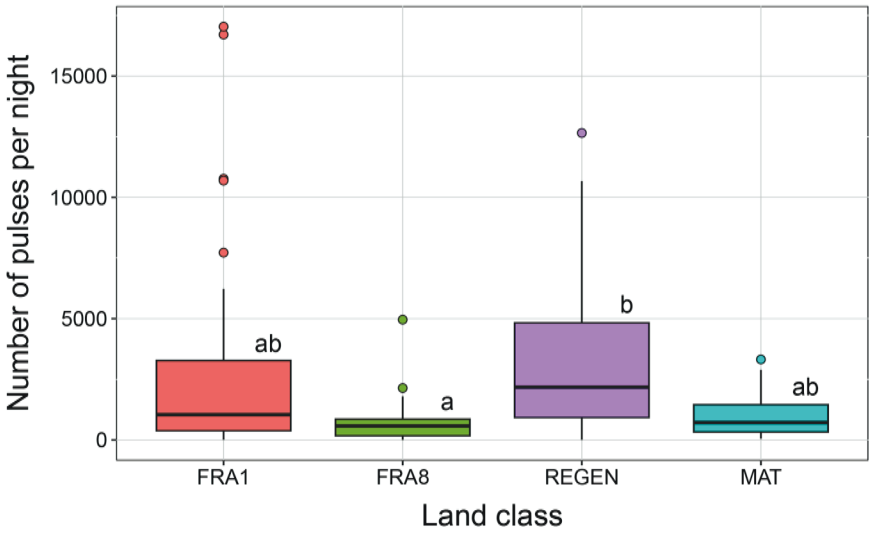

Figure 3. Boxplots showing the distribution of nightly pulse counts in each land class. Letters denote significance difference ($p < 0.05$).

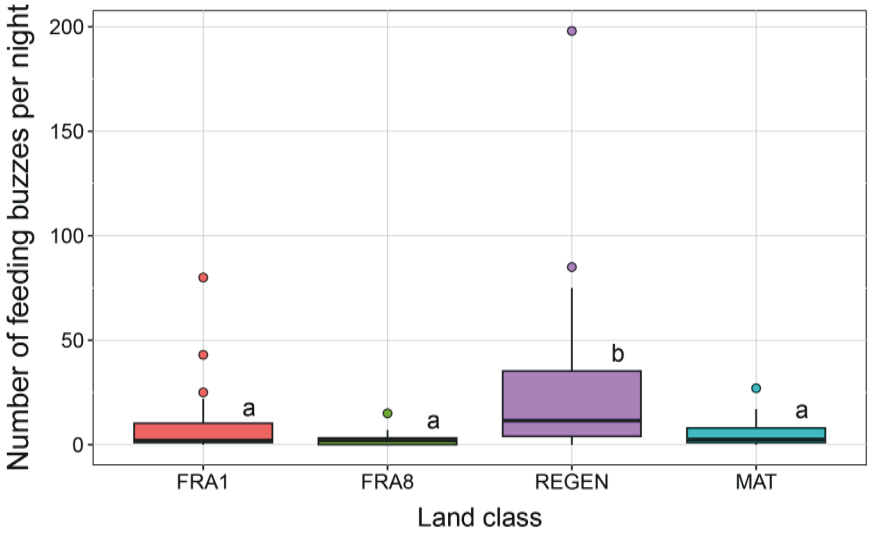

Figure 4. Boxplots showing the distribution of nightly feeding buzzes in each land class. Letters denote significance difference ($p < 0.05$).

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

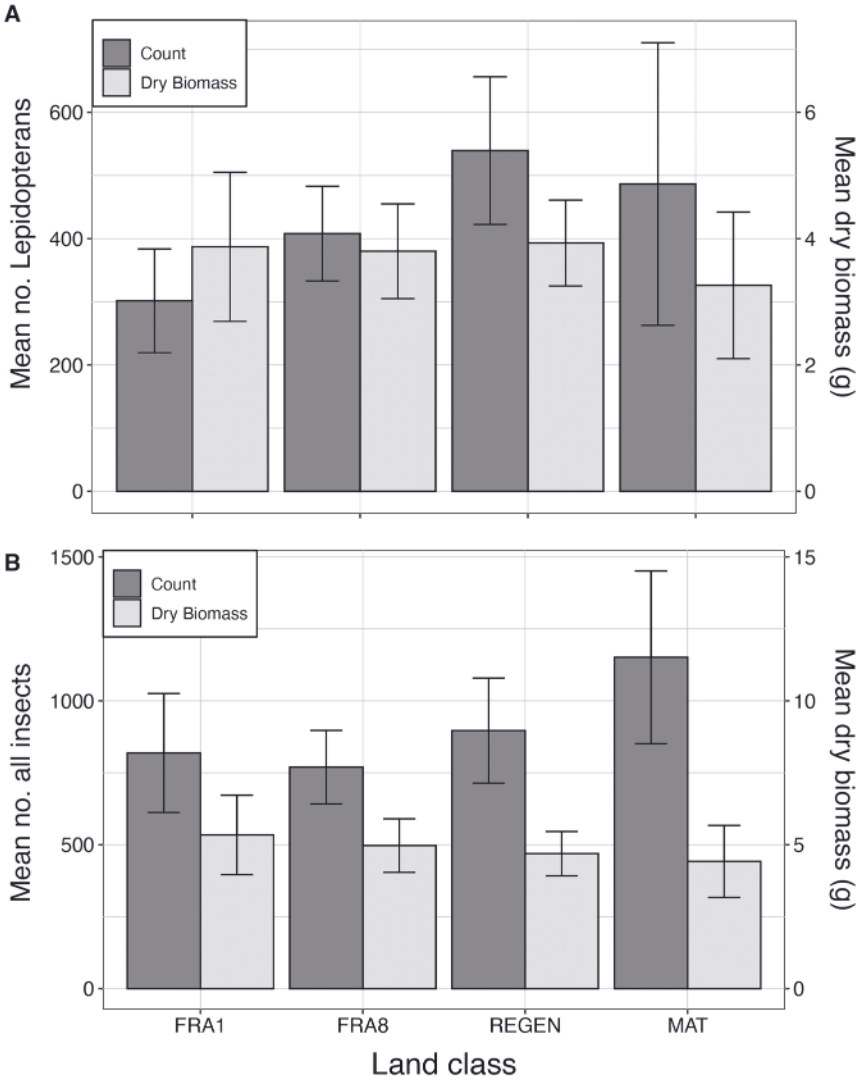

Figure 5. Comparison of Lepidopteran and total insect amounts by land class. A) Mean abundance of Lepidopterans sampled from each land class in counts of individuals and dry biomass (g) including standard error bars. B) Mean abundance of all insects sampled from each land class in counts of individuals and dry biomass (g) including standard error bars. There was not a significant difference in any category.

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 5).

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

Table 3. Summary of habitat parameters for each land class. Ranges and means (n = 4 for all variables except vegetation height which has n = 16) of the measured habitat parameters used in the modeling. The first three variables are characteristics of the pools of water from each site. The last four variables are terrestrial characteristics.

Habitat Parameter		FRA1	FRA8	REGEN	MAT
Pool surface area (m^2)	*Range*	0.69–100.49	6.83–63.36	6.57–72.19	8.50–200.20
	Mean	43.71	39.10	39.93	66.73
Percent cover of pool (%)	*Range*	23.00–48.00	55.00–75.00	0.00–62.00	8.00–75.00
	Mean	26.50	64.00	27.63	38.25
Woody debris volume (m^3)	*Range*	0.00–0.46	0.00–0.22	0.00–0.0005	0.00–2.10
	Mean	0.19	0.10	0.00	0.54
Vegetation height (m)	*Range*	1.10–4.04	1.32–3.60	1.75–13.72	1.23–14.02
	Mean	1.93	2.19	7.78	6.92
Distance to nearest forest (m)	*Range*	28.00–94.00	95.00–131.00	4.00–13.00	2.00–56.00
	Mean	59.00	110.75	9.25	18.25
Percent forest cover (%)	*Range*	0.00–16.50	0.00–7.90	61.80–89.30	53.40–95.00
	Mean	6.70	4.30	75.08	75.58
Distance to nearest road (m)	*Range*	38.00–473.00	90.00–179.00	355.00–504.00	53.00–384.00
	Mean	251.25	144.50	428.50	223.75

Table 4. Mean (standard deviation) of water quality results for each land class. Parameters with (*) denote significant differences ($p < 0.05$) and letters denote the pairwise comparison.

Water Quality Parameter	FRA1		FRA8		REGEN		MAT	
Specific conductivity (umohs/L)	53.87	(43.87)	34.24	(25.96)	21.03	(2.67)	28.51	(18.36)
SO4 (mg/L)	16.01	(8.40)	7.31	(4.88)	7.58	(5.26)	8.30	(6.32)
Mg (mg/L)*	2.68	$(2.15)^{c}$	2.46	$(2.08)^{bc}$	0.63	$(0.15)^{ab}$	0.52	$(0.29)^{a}$
Ca (mg/L)*	4.90	$(5.15)^{b}$	3.43	$(3.38)^{ab}$	0.90	$(0.47)^{a}$	1.86	$(2.18)^{ab}$
K (mg/L)*	3.22	$(2.33)^{b}$	1.47	$(1.34)^{ab}$	0.55	$(0.14)^{a}$	0.53	$(0.21)^{a}$
Na (mg/L)	0.43	(0.22)	0.55	(0.77)	0.34	(0.12)	1.44	(1.99)
Turbidity (NTU)	12.49	(19.39)	5.04	(3.85)	2.03	(1.48)	4.44	(10.18)
pH*	7.01	$(0.69)^{b}$	6.66	$(0.23)^{b}$	5.77	$(0.41)^{a}$	5.88	$(0.44)^{a}$
NO3–N (mg/L)	0.07	(0.07)	0.03	(0.04)	0.05	(0.03)	0.07	(0.08)
Mn (mg/L)*	0.76	$(1.63)^{bc}$	1.89	$(2.81)^{c}$	0.43	$(0.76)^{ab}$	0.07	$(0.03)^{a}$
Fe (mg/L)	0.99	(1.91)	2.47	(5.84)	0.46	(1.00)	1.21	(2.82)

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 5), 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

Table 5. 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 ($\Delta AICc$) 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

Table 5, continued.

Model Sets		K	AIC_c	ΔAIC_c	AIC_w
	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

second-best model, though equally supported (ΔAIC_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 5). 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 5), 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 5) 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).

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 related increase in 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 therefore 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 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. *Lasiurus 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. *Lasiurus 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. *Eptesicus fuscus* will travel and forage in forests, at edges, or in openings, so it was not surprising to detect them in each land class. *Lasiurus 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. *Lasiurus 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 SO_4, 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 has not exhibited toxicity. Additionally, in the Central Appalachians, a specific conductivity threshold of 300 µS/cm is deemed protective of aquatic biota (U.S. EPA 2011), and all 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.

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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References

Agouridis, C.T., P.N. Angel, T.J. Taylor, C.D. Barton, R.C. Warner, X. Yu and C. Wood. 2012. Water quality characteristics of discharge from reforested loose-dumped mine spoil in eastern Kentucky. *Journal of Environmental Quality* 41:454–468.

Agouridis, C.T., C.D. Barton and R.C. Warner. 2018. Recreating a headwater stream system on a valley fill in Appalachia, USA. Pages 147–174 in N.S. Bolan, M.B. Kirkham and Y.S. Ok (eds.), *Spoil to Soil: Mine Site Rehabilitation and Revegetation*. Boca Raton, FL: CRC Press.

Agranat, I. 2014. Detecting bats with ultrasonic microphones: Understanding the effects of microphone variance and placement on detection rates. White paper, Wildlife Acoustics, Inc.

Angel, P., V. Davis, J. Burger, D. Graves and C. Zipper. 2005. The Appalachian Regional Reforestation Initiative. US Office of Surface Mining, Appalachian Regional Reforestation Initiative Forest Reclamation Advisory No. 1.

Barclay, R.M.R. and A. Kurta. 2007. Ecology and behavior of bats roosting in tree cavities and under bark. Pages 17–59 in M.J. Lacki, J.P. Hayes and A. Kurta (eds.), *Bats in Forests: Conservation and Management*. Baltimore, MD: Johns Hopkins University Press.

Barton, C.D., K. Sena, T. Dolan, P. Angel and C. Zipper. 2018. Restoring forests on surface coal mines in Appalachia: A regional reforestation approach with global application. Pages 123–146 in N.S. Bolan, M.B. Kirkham and Y.S. Ok (eds.), *Spoil to Soil: Mine Site Rehabilitation and Revegetation*. Boca Raton, FL: CRC Press.

Biebighauser, T.R. 2003. A guide to creating vernal ponds. U.S.D.A. Forest Service, South Morehead, KY.

Branduzzi, A.M., C.D. Barton and A. Lovell. 2020. First-year survival of native wetland plants in created vernal pools on an Appalachian surface mine. *Ecological Restoration* 38:70–73.

Branduzzi, A.M., C.D. Barton, C.C. Baskin, and A.G. Davis. 2023. Evaluating the use of woody debris to enhance native plant establishment from seeds on legacy coal mines in West Virginia (USA). *Native Plants Journal* 23:272–287.

Brigham, R.M. 2007. Bats in forests: What we know and what we need to learn. Pages 1–15 in M.J. Lacki, J.P. Hayes and A. Kurta (eds.), *Bats in Forests: Conservation and Management*. Baltimore, MD: Johns Hopkins University Press.

Brooks, M.E., K. Kristensen, K.J. van Benthem, A. Magnusson, C.W. Berg, A. Nielsen et al. 2017. glmmTMB balances speed and flexibility among packages for zero-inflated generalized linear mixed modeling. *The R Journal* 9:378–400.

Burles, D., R. Brigham, R. Ring and T. Reimchen. 2009. Influence of weather on two insectivorous bats in a temperate Pacific Northwest rainforest. *Canadian Journal of Zoology* 87:132–138.

Calhoun, A.J.K., M.K. Gahl and R.F. Baldwin. 2012. Northeastern seasonal woodland pools. Pages 135–148 in D.P. Batzer and A.H. Baldwin (eds.), *Wetland Habitats of North America: Ecology and Conservation Concerns*. Berkeley, CA: University of California Press.

Cheng, T.L., A. Gerson, M.S. Moore, J.D. Reichard, J. DeSimone, C.K.R. Willis et al. 2019. Higher fat stores contribute to persistence of little brown bat populations with White-nose Syndrome. *Journal of Animal Ecology* 88:591–600.

Dement, W.T., Z.J. Hackworth, J.M. Lhotka and C.D. Barton. 2020. Plantation development and colonization of woody species in response to post-mining spoil preparation methods. *New Forests* 51:965–984.

Dodd, L.E., M.J. Lacki, D.R. Cox and L.K. Rieske. 2015. Prey consumed by bats across central Appalachia prior to detection of White-nose Syndrome. *Journal of the Kentucky Academy of Science* 75:85–93.

Fenton, M.B. 1990. The foraging behavior and ecology of animal-eating bats. *Canadian Journal of Zoology* 68:411–422.

Frenckell, B. and R.M.R Barclay. 1987. Bat activity over calm and turbulent water. *Canadian Journal of Zoology* 65:219–222.

Frick, W.F., S.J. Puechmaille and C.K.R. Willis. 2016. White-nose Syndrome in bats. Pages 245–262 in C.C. Voigt and T. Kingston (eds.), *Bats in the Anthropocene: Conservation of Bats in a Changing World*. Cham, Switzerland: Springer.

Frick, W.F., T. Kingston and J. Flanders. 2019. A review of the major threats and challenges to global bat conservation. *Annals of the New York Academy of Sciences* 1469:5–25.

Greenberg, A.E., L.S. Clesceri and A.D. Eaton. 1992. *Standard Methods for the Examination of Water and Wastewater*. Washington, DC: American Public Health Association.

Grindal, S.D. and R.M. Brigham. 1999. Impacts of forest harvesting on habitat use by foraging insectivorous bats at different spatial scales. *Ecoscience* 6:25–34.

Hackworth, Z., J. Lhotka, J. Cox, C. Barton, and M. Springer. 2018. First-year vitality of reforestation plantings in response to herbivore exclusion on reclaimed Appalachian surface-mined land. *Forests* 9:222.

Hoyt, J.R., A.M. Kilpatrick and K.E. Langwig. 2021. Ecology and impacts of White-nose Syndrome on bats. *Nature Reviews Microbiology* 19:196–210.

Jantzen, M.K. and M.B. Fenton. 2013. The depth of edge influence among insectivorous bats at forest-field interfaces. *Canadian Journal of Zoology* 91:287–292.

Korine, C., R. Adams, D. Russo, M. Fisher-Phelps and D. Jacobs. 2016. Bats and water: Anthropogenic alterations threaten global bat populations. Pages 215–241 in C.C. Voigt and T. Kingston (eds.), *Bats in the Anthropocene: Conservation of Bats in a Changing World*. Cham, Switzerland: Springer.

Krusic, R.A., M. Yamasaki, C.D. Neefus and P.J. Pekins. 1996. Bat habitat use in White Mountain National Forest. *Journal of Wildlife Management* 60:625–631.

Kurta, A., J.P. Hayes and M.J. Lacki. 2007. *Bats in Forests: Conservation and Management*. Baltimore, MD: Johns Hopkins University Press.

Lacki, M.J., S.K. Amelon and M.D. Baker. 2007. Foraging ecology of bats in forests. Pages 83–127 in M.J. Lacki, J.P. Hayes and A. Kurta (eds.), *Bats in Forests: Conservation and Management*. Baltimore, MD: Johns Hopkins University Press.

Lambert, M., A.N. Drayer, W. Leuenberger, S.J. Price and C.D. Barton. 2021. Evaluation of created wetlands as amphibian habitat on a reforested surface mine. *Ecological Engineering* 171:186306.

Langridge, J., B. Pisanu, S. Laguet, F. Archaux and L. Tillon. 2019. The role of complex vegetation structures in determining hawking bat activity in temperate forests. *Forest Ecology and Management* 448:559–571.

Larkin, J.L., J.J. Krupa, J.J. Cox, K. Alexy, D.E. Unger and C. Barton. 2008. Small mammal response to vegetation and spoil conditions on a reclaimed surface mine in eastern Kentucky. *Southeastern Naturalist* 7:401–412.

Law, B. and M. Chidel. 2002. Tracks and riparian zones facilitate the use of Australian regrowth forest by insectivorous bats. *Journal of Applied Ecology* 39:605–617.

Law, B., K.J. Park and M.J. Lacki. 2016. Insectivorous bats and silviculture: Balancing timber production and bat conservation. Pages 105–150 in C.C. Voigt and T. Kingston (eds.), *Bats in the Anthropocene: Conservation of Bats in a Changing World*. Cham, Switzerland: Springer.

Lenth, R. 2022. emmeans: Estimated marginal means, aka least-squares means. R package version 1.8.1–1. Available at: https://CRAN.R-project.org/package=emmeans.

Lituma, C.M., J.J. Cox, S.F. Spear, J.W. Edwards, J.L. De La Cruz, L.I. Muller and W.M. Ford. 2021. Terrestrial wildlife in the post-mined Appalachian landscape: Status and opportunities. Pages 135–166 in C.E. Zipper and J. Skousen (eds.), *Appalachia's Coal-Mined Landscapes*. Cham, Switzerland: Springer.

Loeb, S.C. and J.M. O'Keefe. 2011. Bats and gaps: The role of early successional patches in the roosting and foraging ecology of bats. Pages 167–189 in C.H. Greenburg, B.S. Collins and F.R. Thompson III (eds.), *Sustaining Young Forest Communities*. Cham, Switzerland: Springer.

Mackey, R.L. and R.M. Barclay. 1989. The influence of physical clutter and noise on the activity of bats over water. *Canadian Journal of Zoology* 67:1167–1170.

Mazerolle, M.J. 2006. Improving data analysis in herpetology: Using Akaike's Information Criterion (AIC) to assess the strength of biological hypotheses. *Amphibia-Reptilia* 27:169–180.

Mazerolle, M.J. 2020. AICcmodavg: model selection and multimodel inference based on (Q)AIC(c). R package version 2.3-1. Available at: https://cran.r-project.org/package=AICcmodavg.

Menzel, J.M., M.A. Menzel, J.C. Kilgo, W.M. Ford and J.W. Edwards. 2005. Bat response to Carolina bays and wetland restoration in the southeastern U.S. coastal plain. *Wetlands* 25:542–550.

Miller, C., G. King, Y. Liu, R. Harrison, E. Turnblom and D. Zabowski. 2015. Assessing Douglas-fir seedling establishment using two modified forestry reclamation approaches in the Pacific Northwest. *Forests* 6:2836–2852.

Moore, L.H. and T.L. Best. 2018. Impact of vegetation on activity of bats over wetlands in coastal South Carolina. *Journal of Mammalogy* 99:1082–1092.

Muncy, B.L., S.J. Price, S.J. Bonner and C.D. Barton. 2014. Mountaintop removal mining reduces stream salamander occupancy and richness in southeastern Kentucky (USA). *Biological Conservation* 180:115–121.

Murray, S.W. and A. Kurta. 2004. Nocturnal activity of the endangered Indiana bat (*Myotis sodalis*). *Journal of Zoology* 262:197–206.

Owen, S.F., M.A. Menzel, J.W. Edwards, W.M. Ford, J.M. Menzel, B.R. Chapman et al. 2004. Bat activity in harvested and intact forest stands in the Allegheny Mountains. *Northern Journal of Applied Forestry* 21:154–159.

Price, S.J., B.L. Muncy, S.J. Bonner, A.N. Drayer and C.D. Barton. 2016. Effects of mountaintop removal mining and valley filling on the occupancy and abundance of stream salamanders. *Journal of Applied Ecology* 53:459–468.

Russo, D., L. Cistrone and G. Jones. 2012. Sensory ecology of water detection by bats: A field experiment. *PLoS ONE* 7:e48144.

Russo, D., L. Ancilloto and G. Jones. 2018. Bats are still not birds in the digital era: Echolocation call variation and why it matters for bat species identification. *Canadian Journal of Zoology* 96:63–78.

Schober, P., C. Boer and L.A. Schwarte. 2018. Correlation coefficients: Appropriate use and interpretation. *Anesthesia & Analgesia* 126:1763–768.

Seibold, S., J. Buchner, C. Bässler and J. Müller. 2013. Ponds in acidic mountains are more important for bats in providing drinking water than insect prey. *Journal of Zoology* 290:302–308.

Sena, K., C. Barton, P. Angel, C. Agouridis and R. Warner. 2014. Influence of spoil type on chemistry and hydrology of interflow on a surface coal mine in the eastern US coalfield. *Water, Air, and Soil Pollution* 11:2171.

Shartitz, R.R., D.P. Batzer and S.C. Pennings. 2014. Ecology of freshwater and estuarine wetlands: An introduction. Pages 1–22 in D.P. Batzer and R.R. Shartitz (eds.), *Ecology of Freshwater and Estuarine Wetlands*. Oakland, CA: University of California Press.

Symonds, M.R.E. and A. Moussalli. 2011. A brief guide to model selection, multimodel inference and model averaging in behavioural ecology using Akaike's information criterion. *Behavioral Ecology and Sociobiology* 65:13–21.

U.S.D.A Forest Service. 2016. Mower Tract restoration and recreation enhancement project. Environmental Assessment. Monongahela National Forest, WV: Greenbrier Ranger District.

U.S. EPA. 2011. A field-based aquatic life benchmark for conductivity in Central Appalachian streams. National Center for Environmental Assessment, Office of Research and Development, U.S. Environmental Protection Agency. Technical Report EPA/600/R-10/023F.

U.S. Fish and Wildlife Service. 1967. Native fish and wildlife; endangered species. Federal Register 32:4001.
U.S. Fish and Wildlife Service. 1976. Endangered and threatened wildlife and plants; determination that two species of butterflies are threatened species and two species of mammals are endangered species. Federal Register 41:17736–17742.
U.S. Fish and Wildlife Service. 1979. Listing of Virginia and Ozark big-eared bats as endangered species, and critical habitat determination; final rule. Federal Register 44:69206–69208.
U.S. Fish and Wildlife Service. 2022. Species status assessment report for the northern long-eared bat (*Myotis septentrionalis*), Version 1.2. Bloomington, MN.
Wilkinson, L.C. and R.M.R. Barclay. 1997. Differences in the foraging behaviour of male and female big brown bats (*Eptesicus fuscus*) during the reproductive period. *Ecoscience* 4:279–285.
Williamson, T.N. and C.D. Barton. 2020. Hydrologic modeling to examine the influence of the forestry reclamation approach and climate change on mineland hydrology. *Science of the Total Environment* 743:140605.
Wolbert, S.J., A.S. Zellner and H.P. Whidden. 2014. Bat activity, insect biomass, and temperature along an elevational gradient. *Northeastern Naturalist* 21:72–85.
Zimmerman, G. and W. Glanz. 2000. Habitat use by bats in eastern Maine. *Journal of Wildlife Management* 64:1032–1040.
Zipper, C.E., J.A. Burger, J.G. Skousen, P.N. Angel, C.D. Barton, V. Davis and J.A. Franklin. 2011. Restoring forests and associated ecosystem services on Appalachian coal surface mines. *Environmental Management* 47:751–765.

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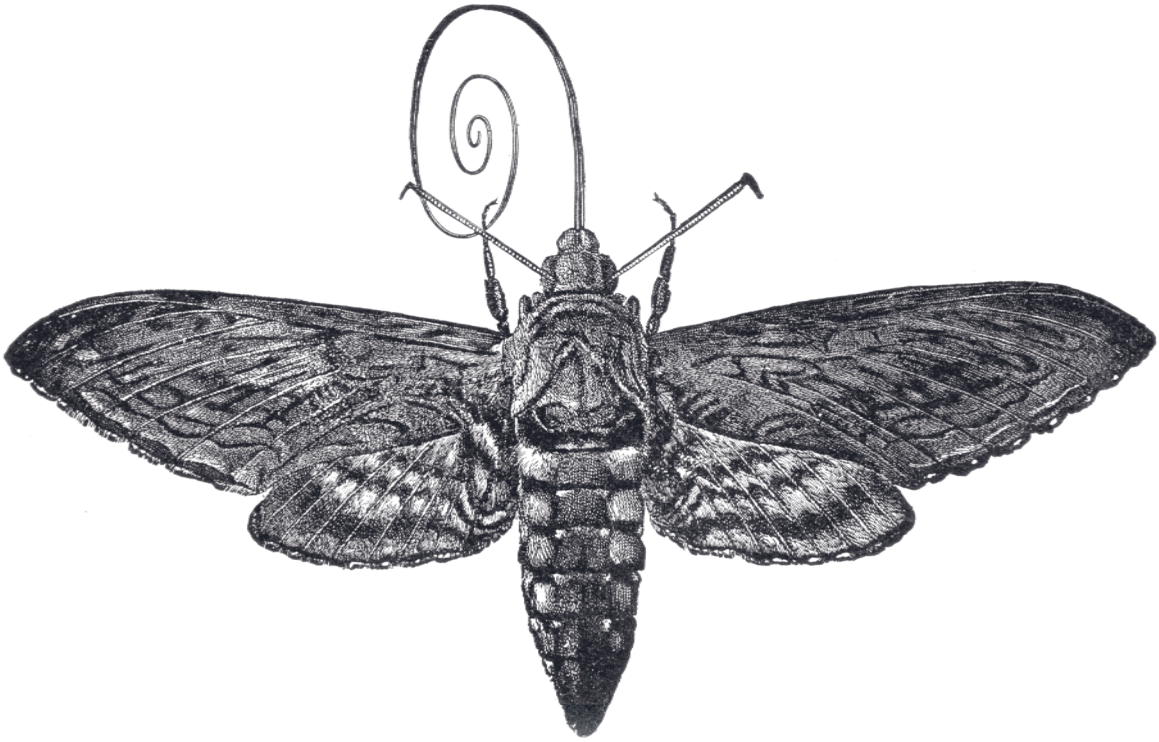

Hawk moth. Source: Buel P. Colton, *Zoology: Descriptive and Practical* (Boston: D.C. Heath & Co., 1903). The Florida Center for Instructional Technology, College of Education, University of South Florida. fcit.usf.edu.