2021 Report

Assessing Eastern whip-poor-will and monarch butterfly responses to NRCS Conservation Programs Targeting Early-successional Habitats in the Eastern Forests

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Introduction

Throughout the eastern deciduous forests of North America, a lack of disturbance coupled with advancing ecological succession in many regions has led to forests dominated by even-aged sawtimber with very little in the early successional stage (Shifley et al 2014; King and Schlossberg 2014). An assessment of age distribution and disturbance legacy of North American forests found that only 8.6% of forest land in the northeastern United States was 0–20 years old (Pan et al., 2011). Limited availability of early successional forests and shrublands has contributed to the declines of many disturbance-dependent species (King and Schlossberg 2014; Livaitis 2001). The implementation of science-based best management practices that create or maintain habitat for the most vulnerable of these species is thought to be an important step toward reversing their population declines. For the past decade, Natural Resource Conservation Service's (NRCS) Working Lands for Wildlife Program in several Appalachian and New England states and a Regional Conservation Partnership Program (RCPP) in the upper Great Lakes states have been working with private landowners to implement conservation practices that create and restore early-successional habitat (ESH) for two species of greatest conservation concern; New England Cottontail (NEC; *Sylvilagus transitionalis*) and Golden-winged Warbler (GWWA; *Vermivora chrysoptera*).

Monitoring of response of these target species to early successional communities created through Working Lands for Wildlife and Regional Conservation Partnership Programs have been completed (i.e., McNeil et al. 2020, Livaitis et al. 2021). The increased availability of these restored early successional communities has potential value for conserving other declining species such as pollinators (Roberts et al. 2017), American Woodcock (Johnson 2020), bats (Wright et al 2021), and ruffed grouse (Dessecker and McAuley 2001), to name a few. As such, understanding the potential benefits of forest management to other at-risk species would provide critical insight to forest managers and conservation biologist who wish to restore forested landscapes that support the recovery of multiple species. This project aims to assess the potential benefits of NRCS conservation programs that target GWWA and NEC to two additional at-risk species associated with to early successional communities; Eastern Whip-poor-will (*Antrostomus vociferus*) and monarch butterfly (*Danaus plexippus*).

Eastern Whip-poor-will is a nocturnal bird that experienced a 69% reduction in its population since 1966. Extensive reduction of young forest habitat is thought to be contributing to the species decline. Active forest management plays an important role in creating and maintaining early successional forests for Eastern whip-poor-wills and other declining early successional bird species (Wilson and Watt 2008). As such, efforts to create habitat for GWWA and NEC have great potential to benefit whip-poor-will. The very specific night-time and lunar phase survey protocol for whip-poor-will challenges efforts to evaluate the potential benefits of NRCS conservation programs to this species. However, new technology, specifically low-cost autonomous recording units (ARUs) called 'Audiomoths' (Hill et al. 2019), now make widespread simultaneous nocturnal surveys possible. ARU-based surveys are efficient at detecting whip-poor-wills throughout key singing periods and can be placed at hundreds of off-road sites in a single season. Such a study design is not possible using human observers and traditional road-based survey protocols.

The monarch is one of the eight national-level species targeted under NRCS's Working Lands for Wildlife. Specifically, WLFW-Monarch works with producers and their conservation

partners in portions of the Midwest and southern Great Plains to improve habitat conditions for this migratory butterfly by implementing conservation practices that increase availability of milkweed and other nectar-rich plants. Participating states comprise the core of the monarch's migration route and breeding habitat, and include Illinois, Indiana, Iowa, Kansas, Missouri, Ohio, Oklahoma, Texas, Minnesota, and Wisconsin. The latter two states are also the focus of an ongoing RCPP to create early successional habitat for the Golden-winged Warbler. Due to similar habitat associations, there is potential that early successional communities restored for Golden-winged Warblers may benefit monarchs. We will use standard monarch butterfly habitat and population monitoring protocols developed by the Monarch Joint Venture's Integrated Monarch Monitoring Program to assess the degree to which the species may benefit from the GWWA-RCPP in Minnesota and Wisconsin.

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Part I: An assessment of monarch butterfly (Danaus plexippus) and other pollinators use of early successional communities managed for golden-winged warbler (Vermivora chrysoptera) in the western Great Lakes

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Abstract: The eastern monarch butterfly population has severely declined over the past 25+ years resulting in a need to stabilize and conserver the population, by increasing milkweed stems and nectar-rich plants throughout the landscape in central and eastern North America. Managing for early successional communities, could provide an opportunity to increase monarch habitat. The purpose of this study is to quantify the degree to which monarch butterflies and other pollinators benefit from early successional habitat management for the golden-winged warbler, enrolled within USDA Natural Resources Conservation Service (NRCS) Regional Conservation Partnership Program, in the northern Great Lakes. In June-August 2021, we surveyed for pollinators and floral resources within two early successional community types (alder shrub-sheard (hereafter "alder") n = 24, upland young forests (hereafter "upland"), n = 25) managed for the Golden-winged Warbler in northern Minnesota and Wisconsin. We compared the presence of monarchs and milkweed between upland and alder sites and reference sites that were monitored by the Monarch Joint Venture's Integrated Monarch Monitoring Program (n = 25). We found that reference sites had a higher presence of monarchs (96%, 24/25) compared to alder (54%, 13/24) and upland sites (60%, 15/25). Between alder and upland sites, we found that milkweed density, floral abundance, and study sites with a higher proportion of herbaceous wetlands in the nearby landscape were consistent predictors for monarch and milkweed presence. Lastly, we also found abundant floral resources positively predicted bumble bee density, whereas relationships were less clear for butterflies. Our *preliminary* results support that increasing milkweed and nectar-rich plants within alder and upland early successional communities will provide valuable resources for the monarchs and other pollinators in the northern Great Lakes.

Introduction

It is estimated that the eastern monarch (*Danaus plexippus*) population has declined about 80% over the past 25+ years (Boyle et al., 2019). If these rates are left unchecked, researchers estimate that there is a 11-57% chance that monarch populations may not be recoverable in 20 years (Semmens et al., 2016). In response, there has been urgency to protect and conserve the eastern monarch population, including the recent decision to be wait listed for the Endangered Species Act (USFWS, 2020). The loss of breeding habitat and degradation (Thogmartin et al., 2017) are thought to currently be the greatest cause of decline. Conservation planners have set goals to increase the eastern monarch population to 225 million individuals and overwintering occupancy area of 6 hectares by 2020 (Pollinator Health Task Force, 2015). To achieve this goal, it has been suggested that 1.3–1.6 billion new milkweed stems will need to be established over the next decade throughout a twenty-state portion of the species North American breeding range (Pleasants, 2017). Managing for early successional communities could provide an opportunity to increase availability of milkweed and other nectar-rich plants, the two most important habitat components for monarchs.

Early successional communities provide important resources for pollinators (Hanula et al., 2016; Mathis et al., 2021). In general, bees and butterflies benefit from early successional communities, created by different forest management practices, because of an abundance of nectar plants (Roberts et al., 2017), warm microclimates (Korpela et al., 2015), bare soil for nesting bees (Proctor et al., 2012), and host plants for butterflies (Fartmann et al., 2013). A study in the Ozarks of west-central Arkansas found that thinning and frequent prescribe fire restoration of shortleaf pine savannah communities had higher abundances of nectar resources and monarchs during the fall migration (Rudolph et al., 2006). Therefore, investigating whether early successional communities provide resources, such as nectar plants and milkweed, for monarchs could expand conservation efforts.

The monarch is one of eight national-level species targeted under the Natural Resources Conservation Service's (NRCS) Working Lands for Wildlife (WLFW) initiative. Two participating states are Minnesota and Wisconsin, which are also the focus of an ongoing NRCS Regional Conservation Partnership Program (RCPP) to create early successional habitat for the Golden-winged Warbler (*Vermivora chrysoptera*) (McNeil et al., 2020b). Due to similar habitat associations, there is evidence that early successional communities managed for Golden-winged Warblers in the Appalachian Mountains benefits native bees and butterflies (Mathis et al., 2021; Lee et al., 2021). The purpose of this study is to quantify the degree to which monarch butterflies and other pollinators (butterflies and bumble bees) benefit from NRCS-RCPP early successional habitat management for the Golden-winged Warbler in the western Great Lakes.

Objectives

1. Compare monarch adult, egg, larvae, and milkweed counts and densities among three managed early successional community types (alder shrublands, upland forets), and herbaceous-dominated reference sites managed for pollinators (here after reference sites), using summary statistics.
2. Compare monarch adult, egg, larvae, and milkweed presence between alder and upland sites. Additionally, identify what features at a landscape and local scale best predict presence of monarch adult, egg, larvae, and milkweed.
3. Compare all butterfly and bumble bee densities between alder and upland sites. Also, identify what features at a landscape and local scale best predict densities of these two groups.

Methods

Study Area

This study was conducted in northern Minnesota and Wisconsin at approximately 44 – 49°N and 89 - 97°W and elevation range 249-540 meters (Figure 1). We selected 49 study sites based on the following criteria: 1) majority of sites enrolled within NRCS-RCPP for GWWA management, 2) half were alder shrub-sheard (n=24) and half were upland young forest (n=25) 3) management was conducted 1-5 years prior to monitoring, 4) managed area was ≥ 2.6 ha, and 5) sites of the same management type were at least 700 meters apart (McNeil et al., 2019). GWWA managed sites ranged in size from 2.6 – 52.2 ha (n=49, mean: 10.1 ha, median: 7.6 ha, SD: 8.6 ha). Most sites were privately owned (n=39). Many private sites were enrolled in NRCS's Regional Conservation Partnership Program (RCPP). Other programs that funded the

management on privately owned sites are Fish and Wildlife Service Partners Program (FWSPP), Environmental Quality Incentives Program (EQIP), and Great Lakes Restoration Initiative (GLRI). Public sites were located on lands managed by the state of Minnesota (Cloquet Area Forest), or Carlton, Aikin, Douglas, and Saint Louis Counties.

For each study site, we created a 1-ha monitoring plot, within which all surveys were conducted. To create monitoring plots, we followed the Monarch Joint Venture's Integrated Monarch Monitoring Program (hereafter IMMP) (MJV, 2020). We visited each study site three times from June 2nd – August 26th, 2021. Five surveys: (1) pollinator, (2) milkweed, (3) monarch egg and larvae, (4) blooming nectar plants, and (5) structural vegetation were conducted within each 1-ha monitoring plot. For surveys 1-4, we followed the IMMP and these were conducted during all three visits. The structural vegetation survey was modified from a previous study (see below) and was only conducted during the second visit in July (McNeil et al., 2019).

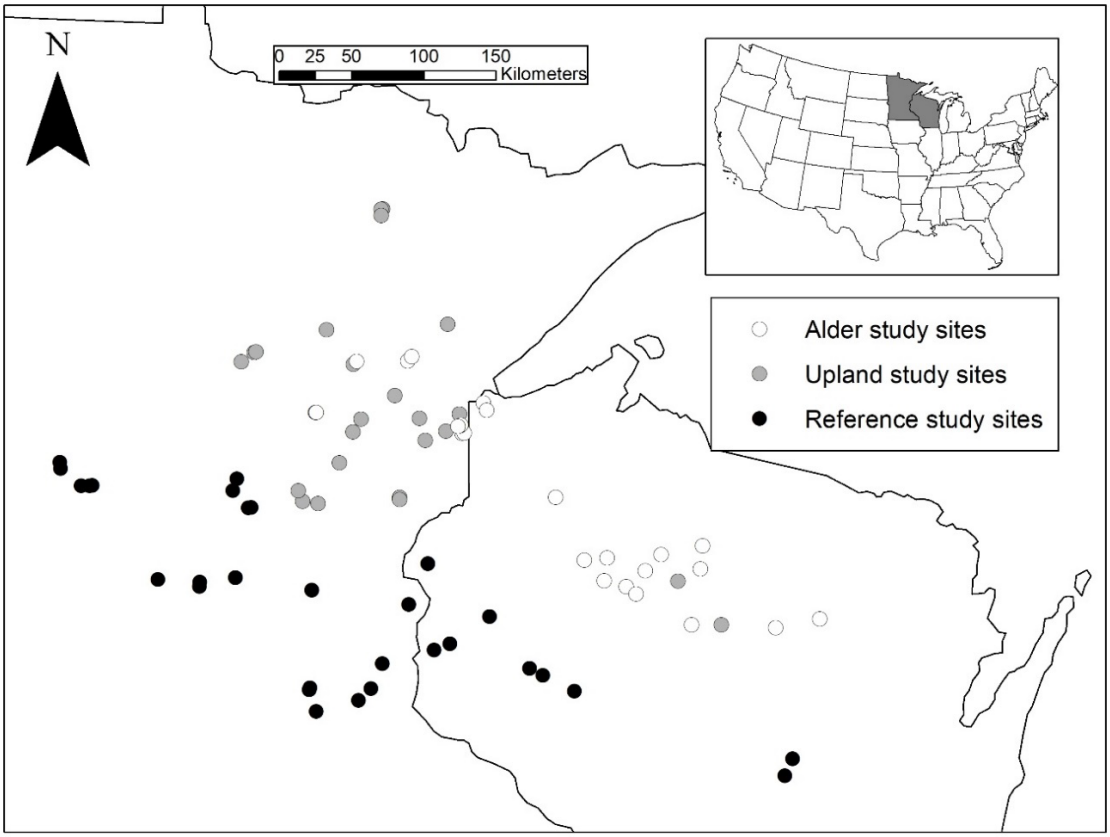

Figure 1. Distribution of study sites throughout northern Minnesota and Wisconsin where pollinator communities were surveyed. Two types of managed early successional communities, alder shrub sheared (n=24, white circles) and upland young forests (n=25, gray circles), were investigated from June-August 2021. Reference study sites (black circles) were acquired from the Monarch Joint Venture Integrated Monarch Monitoring Program from 2020 and 2021. Note: locations of private sites are randomly shifted to protect landowner privacy.

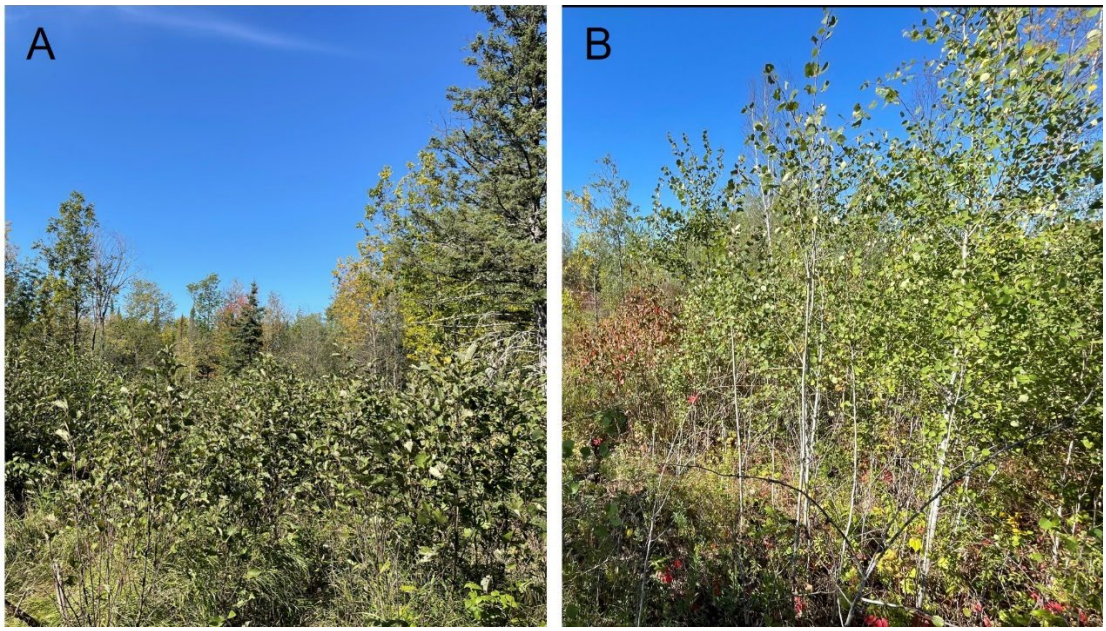

Photo set 1. Representative photos of the two managed early successional communities in northern Minnesota and Wisconsin within which we surveyed pollinator communities. Specifically, two types of managed early successional communities were investigated from June-August 2021, A) alder shrub sheared and B) upland young forests.

Pollinator survey

Upon arrival to a site, a 25-minute timed visual pollinator survey was completed first to avoid flushing or disturbing monarchs and other pollinators during other activities. The purpose of this survey was to count adult monarchs, all butterflies, and bumble bees using distance sampling (McNeil et al., 2019). Resulting data were used to estimate pollinator density and account for imperfect detection. To conduct the survey one individual walked the perimeter of the monitoring plot at a consistent rate of 1m/3sec (rectangle and square: distance = 500 meters, time = 25 minutes; if irregular plot: time varied depending on perimeter length). For each pollinator observed, we recorded perpendicular distance from transect at first detection, behavior, and any associated plant species if a pollinator was nectaring or a monarch was oviposting. All pollinator surveys were conducted between 10 am – 5 pm, ≥ 15.6 °C, and restricted to days with no rain or high winds (Beaufort wind code ≤ 5).

Milkweed survey and Monarch egg and larvae survey

We conducted the milkweed and monarch egg and larvae surveys simultaneously with the pollinator survey. If milkweed was detected we recorded species, perpendicular distance from transects, number of plants, and number of stems. Additionally, if milkweed was within five meters of the transect, we stopped the timer and carefully searched the entire plant for monarch eggs and larvae. If milkweed was beyond five meters we would record the same information, but not search it for monarch eggs and larvae. Only queen butterflies (*Danaus gilippus*) have eggs that look like those of monarchs (MJV, 2020), however this species does not occur in the western Great Lake states.

We calculated monarch egg and larvae density per m^2 by multiplying monarch egg or larvae abundance per milkweed stems searched by milkweed stem density (milkweed stems/m^2). Milkweed density (milkweed stems/m^2) was calculated by multiplying the number of milkweed plants and stems, then dividing by area searched (m^2). We followed this method of calculating monarch eggs and larvae density because that is how densities for the MJV reference sites were calculated (Bruce et al., 2021) and this allowed us to compare abundance of monarch eggs and larvae with an area-based metric.

Blooming nectar plant survey

A blooming nectar plant survey was conducted to quantify available floral resources (flowers/m^2) during each visit. Multiple transects (transect 1 = 200m, transect 2 = 200 m, transect 3 = 100m, total = 500m) were established within the monitoring plot following the IMMP (MJV, 2020). A 1-m^2 subplot was placed every 5 meters along the transects (transect 1 = 40 subplots, transect 2 = 40 subplots, transect 3 = 20 subplots) for a total of 100 subplots. To reduce vegetation disturbance resulting from visiting each site three times, we offset the subplot by one meter from the transect during each visit (e.g., first visit – subplots centered over center of transect, second visit – subplots moved one meter to left, third visit – subplots moved one meter to right). Within each subplot, we counted number of stems with flowers and number of individual flowers on each stem. We counted individual stems and flowers if < 20 and estimated to the nearest 10 if there were > 20 stems or flowers/stem. We were not able to complete the same number of subplots during each visit. Therefore, to account for this we divided total number of flowers during a single visit by number of subplots completed during that visit to acquire flowers/m^2.

We calculated percentage of subplots with a blooming nectar plant present, species richness, and floral resources abundance for explanatory variables for our analysis. The percentage of subplots with a blooming nectar plant present was calculated by determining how many subplots had a blooming plant recorded divided by total number of subplots conducted. Species richness was calculated using *specnumber* function within *vegan* package in R (R Core Team, 2020). To calculate floral abundance, we log transformed flowers/m^2. Lastly, we calculated site's average across three survey rounds for these three variables.

Structural vegetation survey

We conducted a structural vegetation survey within each study site during the second visit in July. This survey was modified from Mathis et al., (2021). We used an ocular tube to quantify percent cover of 12 different structural vegetation strata (Table 1). There were 25 stops/site, and each stop was spaced 20m apart. At each stop, we recorded those vegetation strata that intersected with the ocular tube crosshairs. Percent cover was calculated for each of the 12 vegetation strata categories by dividing number of intersections by number of stops (25) and multiply by 100. Lastly, we used a 10-factor wedge prism and counted number of "in" and "half" trees at the corners and center of the monitoring plot (n=5 prism readings). Prism data were used to calculate study site tree basal area (m^2/ha).

The Monarch Joint Venture IMMP follows a simpler method to quantify structural vegetation within the monitoring plot that we also recorded. During the first survey round, after walking throughout the monitoring plot, we estimated percent of plot covered by grass, forbs, shrubs, and trees to 5 different bins (0%, 1-10%, 11-25%, 26-50%, >50%).

Table 1. Definitions of 12 different vegetation strata that were recorded during the structural vegetation survey. This survey was only conducted during the second sampling round in July 2021 and these data were only collected for the alder and upland study sites located in northern Minnesota and Wisconsin.

Vegetation Strata Category	Definition
Canopy	living tree (snags are not included) with a DBH ≥10 cm.
Large sapling	young tree (<10 cm DBH), woody and generally has one main stem, that is taller than 1m
Small sapling	young tree (<10 cm DBH), woody and generally has one main stem, that is shorter than 1m
Large shrub	woody and generally with multiple stems that branch below the soil, that is taller than 1m
Small shrub	woody and generally with multiple stems that branch below the soil, that is taller than 1m
Rubus spp.	*Rubus* spp. (e.g., raspberry or blackberry), brambles with single or multiple canes with thorns.
Forb	Broad category for herbaceous plants that are dicots
Fern	Any species of fern, which have feathery or leafy fronds and reproduce by spores
Coarse Woody Debris	Any woody debris laying on the ground and is ≥10 cm diameter
Grass	Any grass or sedge, also known as monocots
Leaf Litter	Dead leaves or vegetation from trees, forbs, grass, and small twigs or branches <10 cm diameter
Bare ground	Bare soil. Could not have both leaf litter and bare ground, only presence of one or the other.

Landscape Variables

We were also interested if the surrounding landscape influenced presence or density monarch butterflies and other pollinators, therefore we calculated the percent area of six different land cover categories extracted from the 2019 National Land Cover Database (NLCD 2019, n.d., 30m resolution). The six NLCD categories we collected were (41) Deciduous Forest, (43) Mixed Forest, (71) Grassland/Herbaceous, (81) Pasture/Hay, (90) Woody Wetlands, and (95) Emergent Herbaceous Wetlands, because these are the main types of land cover within the Golden-winged Warbler Conservation Region. To complete this we used the *extract* function within the *raster* package in R (R Core Team, 2020). We extracted percent area within a 10 and 2-km radius of the monitoring plot centroid. We used a 10km radius because it is estimated to be the maximum daily travel distance for a female monarch (Bruce et al., 2021; Zalucki, 2016) and we used a 2km

radius because this is a scale that landscape cover type has been assessed for other pollinators (McNeil et al., 2020a) and it is about the scale used for Golden-winged Warbler management. Although, the percent area at 10km and 2km for each of the six land cover categories were highly correlated (r >0.70). Thus, for analysis we only included the variables extracted at 2km, because the primary goal of this study is to assess pollinator use within the context of sites management Golden-winged Warbler.

Selection of Monarch Joint Venture Data

We acquired data for the reference sites from the Monarch Joint Venture's Integrated Monarch Monitoring Program (IMMP) to statistically compare monarch butterfly use of habitats that are managed for the Golden-winged Warbler (alder and upland) to sites that are specifically managed for monarchs and other pollinators. For the reference sites, we selected surveys from years 2020 and 2021 (see more below). Before running any statistical tests or calculating summary statistics, we created certain criteria to filter the reference sites, so these sites were comparable with the alder and upland sites, such as survey conducted, location, and sampling effort. For the reference sites we only included sites that were: (1) > 700 meters from another study site, (2) comparable site types ("Protected Grassland", "Unclassified Grassland", "Agricultural Conservation Lands", and "Rights-of-way"), in other words we excluded all "Developed" and "Row Crop" site types because we believed these disturbance intensity on such sites would not appropriate to compare with the alder and upland study sites (Site Types referenced from IMMP protocol), (3) followed the IMMP 1-ha monitoring plot protocol (removed all "census surveys"), (4) and within USDA Level III Ecoregions 50 (Northern Lakes and Forests) and 51 (North Central Hardwood Forests) (Ecoregions, n.d). All alder and upland study sites were within ecoregions 50 and 51, therefore this was used as a location-based boundary for the reference sites to avoid comparing study sites in different landscapes, which could result in confounding factors. Next, we restricted our use of data from reference sites to those collected within the same survey period as our alder and upland sites (June 2nd – August 26th) and followed the same adult monarch survey criteria (10am-5pm, ≥60°F (15.6°C), wind speed ≤ 5 Beaufort scale, and no rain).

Lastly, we created a selection process to make the sampling effort between the reference and alder and upland sites similar (number of visits = 1-3). The reference study sites had a range of 1-17 visits, whereas as alder and upland study sites always had 3 visits. First, all surveys types (adults, milkweed, egg/larvae) had to be completed within three days of each other to be considered a unique visit (10/55 selected visits were ≤3 days apart). Second, all visits within the three survey rounds (round 1 = June 2nd – June 30th, round 2 = July 1st – July 29th, round 3 = July 30th – Aug 26th) were ranked based on which survey (adult monarch, monarch egg and larvae, and blooming nectar plant) and how many surveys were conducted to prioritize visits with an adult monarch survey or the most surveys conducted. High to low priority ranks are: 1 = all surveys conducted, 2 = adult monarch survey and one other conducted, 3 = only adult monarch survey conducted, 4 = adult monarch survey conducted. If there was one visit per site, we selected whichever visit was available regardless of the rank. If there were multiple visits per site, and only one visit with a high rank, we selected that visit. If there were multiple visits per

site and multiple visits with the same high rank, we randomly selected one visit. As a result, 22 and 15 unique reference sites were identified from 2020 and 2021, respectively. Eight sites were sampled both years. Thus, we removed these eight study sites from 2020 to avoid overrepresenting a study site. In total, we used data from 29 reference study sites (2020 = 14, 2021 = 15) to compared with alder and upland sites (Table 2).

Table 2. Summary of number of study sites (left side) and total number of visits (sampling effort, right side) for reference, alder, and upland sites by the three different surveys. All study sites were within northern Minnesota and Wisconsin, reference sites were sampled during June-August 2020 and 2021 and alder and upland sites were sampled during June-August 2021. For the left side of the table, the numbers within the table are how many study sites there are for number of visits and for each of the three surveys. For example, reference sites and adult monarch survey, one study site had none (zero visits), 14 study sites had one visit, three study sites had two visits, and 11 study sites had three visits. For the reference sites, each column on the left side has a total of 29, because 29 study sites met our filter criteria (described above). We collected data at 24 alder sites for all three surveys and 25 upland sites for all three surveys. For the right side of the table, the numbers within the table are the total number of visits conducted for each of the three surveys. For example, reference sites and adult monarch survey there was a total of 53 visits conducted. The total number of visits vary for the reference sites between the three different surveys, because all three surveys were not always conducted during each visit, or a survey may have been excluded due to it not meeting our above filtering criteria. For the alder and upland sites, all three surveys were conducted during each visit, therefore the number of visits for each survey is the same.

	# of Visits	Adult monarch survey	Monarch egg and larvae survey	Milkweed survey	Adult monarch survey	Monarch egg and larvae survey	Milkweed survey
Reference sites	0	1	4	4	53	35	35
	1	14	17	17			
	2	3	6	6			
	3	11	2	2			
Alder sites	0	-	-	-	72	72	72
	1	-	-	-			
	2	-	-	-			
	3	24	24	24			
Upland sites	0	-	-	-	75	75	75
	1	-	-	-			
	2	-	-	-			
	3	25	25	25			

Analyses

Comparison of alder, upland, and reference sites

We used reference site data collected in 2020 and 2021(n = 29 sites) to have similar samples sizes to compare with alder (n = 24) and upland (n = 25) sites. From a preliminary analysis we ran Mann Whitney U tests to compare adult monarch counts (average per study site),

monarch eggs per m^2 (average per study site), and monarch larvae per m^2 (average per study site) between the two years for the reference sites. We found that the average number of adult monarchs (W=105, p=0.47) and average number of monarch larvae per study site (W=106, p=0.13) did not differ between years for the reference sites. Although, monarch eggs (W=142, p=0.0005) were statistically different between both years. Therefore, we included survey year as an explanatory variable in future logistic regression models.

We calculated summary statistics to compare the percentage of monarchs present and explanatory variables among the three management types (alder, upland, reference). In the future, we will run logistic regression models to determine what landscape and local habitat characteristics best predict monarch presences among the three management types.

Logistic Regression Analysis – alder and upland sites

As a preliminary analysis, we ran logistic regression models comparing alder and upland sites to determine what landscape and local habitat characteristics best predict monarch presence. We ran models independently for adult monarchs, monarch eggs only, monarch larvae only, all monarch stages combined, and milkweed. The response variable was binary (1=presence, 0=absence). All continuous variables were scaled to have a mean of 0 and standard deviation of 1 using the *scale* function in R (R Core Team, 2020).

We created univariate models within four model sets (Table 3) to test what landscape and local habitat characteristics predicted the response variables. We used the *glm* function in R (R Core Team, 2020). Within each model set we created univariate models for each explanatory variable and compared to a null model. We used Akaike's Information Criterion adjusted for small sample size (AIC_c; Burnham and Anderson, 2002) to rank and assess models. We considered models that had an $AIC_c > 2.00$ ΔAIC_c from a null model and β 95% confidence intervals not including zero (Burnham and Anderson, 2002; Arnold, 2010) to influence or be related to monarch presence at a study site. We assessed model fit by calculating the Brier's score and area under the curve (AUC) by using the *BrierScore* and *roc* functions respectively (R Core Team, 2020).

Table 3. Four model sets were created using a logistic regression to assess landscape and site characteristics that predict monarch and milkweed presence. Model set 1 tests management type (alder or upland) against a null model. Model set 2, univariate models for six NLCD-derived landscape variables. Model set 3 includes site level floral characteristics. Average percentage of subplots with a blooming plant present (Avg NP subplots), average blooming plant species richness (Avg NP richness), average milkweed stems per m^2 (Avg milkweed stems/m^2), and if milkweed was present or absent (milkweed pres.). Lastly, model set 4 includes local scale structural vegetation characteristics. Percent of plot covered by grass, forbs, shrubs, and trees (0%, 11-25%, 26-50%, >50%).

Model Set	Explanatory Variables
Model Set 1	Management type
Model Set 2	Latitude, Longitude, elevation (m), grassland, hay, herb. wetlands, woody wetlands, deciduous forest, mix forest
Model Set 3	Avg NP subplots, Avg NP richness, Avg milkweed stems/m^2, milkweed present
Model Set 4	Percent cover of grass, forbs, shrubs, or trees

Hierarchical Distance Model Analysis

We used Hierarchical Distance Models (HDM) using *unmarked* package in R (R Core Team, 2020; Kéry and Royle, 2015) to estimate the density of butterflies and bumble bees while accounting for imperfect detection (McNeil et al., 2018). We ran these models on our butterfly and bumble bee data independently and were interested in how structural vegetation and floral characteristics impacted butterfly and bumble bee densities across and within the growing season. For the butterfly data set we binned the data into 5 bins: 0-2 m, 2-4 m, 4-6 m, 6-8 m, 8-10m and the bumble bee data was also binned into 5 bins: 0-1 m, 1-2 m, 2-3 m, 3-4 m, 4-5 m. Both datasets the outer 10% of the observations were excluded due to low detection probability (Buckland et al., 2005; McNeil et al., 2019). All continuous covariates were scaled to have a mean of 0 and standard deviation of 1.

First, we created across season HDM to quantify butterfly and bumble bee densities that allow temporary emigration (Kery and Royle, 2015). For both butterfly and bumble bee datasets we ran a landscape and local-scale analysis separately (Table 4). These models are a four-step building process: (1) determine best detection key function, (2) covariates on detection probability (ρ), (3) lambda models (λ) with management type (alder or upland), and (4) lambda models (λ) with local-scale or landscape-scale covariates (Table 4). We compared Akaike's Information Criterion adjusted for small sample size (AIC_c) to assess models. We considered a model significant if it had an $AIC_c > 2.00$ ΔAIC_c from the null model. Lastly, we ran a goodness of fit test on the top model to determine if it was a good fit ($\hat{c} \leq 1.0$). All our models were overdispersed ($\hat{c} > 1$), therefore we compared $\Delta QAIC_c$ to rank models, which accounts for overdispersion.

Table 4. Covariates used to create HDM both across and within season to compare butterfly and bumble bee densities. We used a four-step building process for both across and within season HDM. We included all key functions (exponential, hazard, and halfnormal) and the negative binomial distribution (NB) for each key function combination. To assess detection probability we used all observation covariates, cloud cover (0, 25, 50, 75, 100%), start time of pollinator survey, temperature (°C), wind speed ("low" = (Beaufort scale 0-2 and "high" = Beaufort scale 3-5), date of survey (Julian date), observer, and the 12 structural vegetation classes and stand basal area, because certain vegetation classes can impede observation of pollinators (e.g., high % cover of tall saplings). For the lambda models, first we tested if one of the two management types (alder vs upland) significantly impacted pollinator density. Second group of lambda models, include stand age (number of growing seasons since management), percent cover of the 12 structural vegetation classes, stand basal area, site average floral abundance (Avg ln(flowers/m^2), site average floral species richness (Avg NP richness), and site average percent of subplots with a blooming nectar plant present (Avg NP subplots). The second group of lambda models used in the landscape analysis include elevation (m), latitude, longitude, and the six 2019 NLCD categories.

Key Function	**Detection Probability Models (ρ)**	**Lambda Models (λ) Management Type**	**Lambda models (λ): Site-scale**	**Lambda models (λ): Landscape-scale**
exponential	Cloud cover	Management type	stand age	Elevation (m)
hazard	Start time		canopy cover (%)	latitude
halfnormal	Temperature (°C)		large sapling cover (%)	longitude
exponential (NB)	Wind speed		small sapling cover (%)	Grassland (NLCD code 71)
hazard (NB)	Julian Date		large shrub cover (%)	Pasture/Hay (NLCD code 81)
halfnormal (NB)	Observer		small shrub cover (%)	Herbaceous Wetlands (NLCD code 95)
	canopy cover (%)		rubus cover (%)	Deciduous Forest (NLCD code 41)
	large sapling cover (%)		forb cover (%)	Mixed Forest (NLCD code 43)
	small sapling cover (%)		fern cover (%)	Woody Wetlands (NLCD code 90)
	large shrub cover (%)		grass/sedge cover (%)	
	small shrub cover (%)		cwd cover (%)	
	rubus cover (%)		leaf litter cover (%)	
	forb cover (%)		ground cover (%)	
	fern cover (%)		stand basal area (m^2/ha)	
	grass/sedge cover (%)		Avg ln(flowers/m^2)	
	cwd cover (%)		Avg NP richness	
	leaf litter cover (%)		Avg NP subplots	
	ground cover (%)			
	stand basal area (m^2/ha)			

Second, we created within a growing season HDMs to quantify butterfly and bumble bee densities, which does not allow for temporary emigration (Kery and Royle, 2015). Therefore, we created a model set for each sampling round (n = 3). The same model building process was used for each model set as above (Table 4); we used the respective round value for floral abundance (ln(flowers/m^2)), floral species richness (NP richness), and average percentage of subplots with a blooming nectar plant present (NP subplots), and we did not include landscape-scale variables. We followed the same model selection process as described above.

Preliminary Results

Adult Monarch Butterfly Counts

Of the 29 reference sites, 28 had an adult monarch survey conducted. Adult monarchs were present at 75% (21/28) of the reference sites. During the adult monarch survey at total of 43 monarchs were observed in 2020 (3.1±4.0 (mean±SD) monarchs per site, range = 0-13) and a total of 53 were observed in 2021 (3.5±3.7 (mean±SD) monarchs per site, range=0-11). At the alder and upland sites a total of 30 adult monarchs were observed. Adult monarchs were present at 37% (9/24) of alder sites and 44% (11/25) of upland sites. On average there were 0.5±0.7 (mean±SD) monarchs per site (range=0-2) at alder sites and 0.8±1.0 (mean±SD) monarchs per site (range=0-3) at upland sites.

Monarch Egg and Larvae Counts

Of the 29 reference sites, 25 had monarch egg and larvae survey conducted (n = 25). Monarch eggs were present at 80% (20/25) and monarch larvae were present at 68% (17/25) of reference sites. Density of monarch eggs/ha were higher in 2020 reference sites (40±40 (mean±SD); range = 0-140) compared to 2021 reference sites (2±4, range = 0-3), whereby density of monarch larvae/ha was 9±13 (range = 0-48) and 3±6 (range = 0-21) in 2020 and 2021 respectively. At alder sites, the density of monarch eggs/ha was 0.17±0.46 (range=0-2) and density of monarch larvae/ha was 0.17±0.41 (range=0-1). At upland sites, density of monarch eggs/ha was 0.39±0.11 (range=0-5) and density of monarch larvae/ha was 0.025±0.008 (range=0-0.3). Lastly, monarchs (seen at any stage, adult, egg, larvae) were present at 96% (24/25) of reference sites 54% (13/24) alder sites, and 60% (15/25) upland sites.

Milkweed Counts

Of the 29 reference study sites, 25 had milkweed surveys conducted (n = 25). Milkweed was present at 100% (25/25) of reference sites. On average the density of milkweed stems/ha at reference sites was 400±540 (mean±SD, range=3-2,160). Upland sites had a higher presence of milkweed 36% (9/25), compared to alder sites 29% (7/24). On average the density of milkweed stems/ha was 1.1±4.1 (range=0-20) at alder sites and 1.1±4.5 (range=0-20) at upland sites.

Vegetation and Landscape Characteristics

Landscapes surrounding alder and upland sites were dominated by woody wetlands and deciduous forest, whereas pasture/hay and deciduous forest dominated landscapes surrounding reference sites (Fig. 2). At the site level, reference sites were primarily composed of grass and forbs (Fig. 3), more than half of the subplots had a blooming nectar plant present, and on average

sites had about ten species of blooming nectar plants (Table 5). Alder sites were primarily composed of shrubs and secondarily composed of grass, while upland study sites were largely composed of trees (Fig. 3). Less than 40% of the subplots had a blooming nectar plant present for alder and upland sites and on average there was about seven and ten species of blooming nectar plants at alder and upland sites, respectively (Table 5).

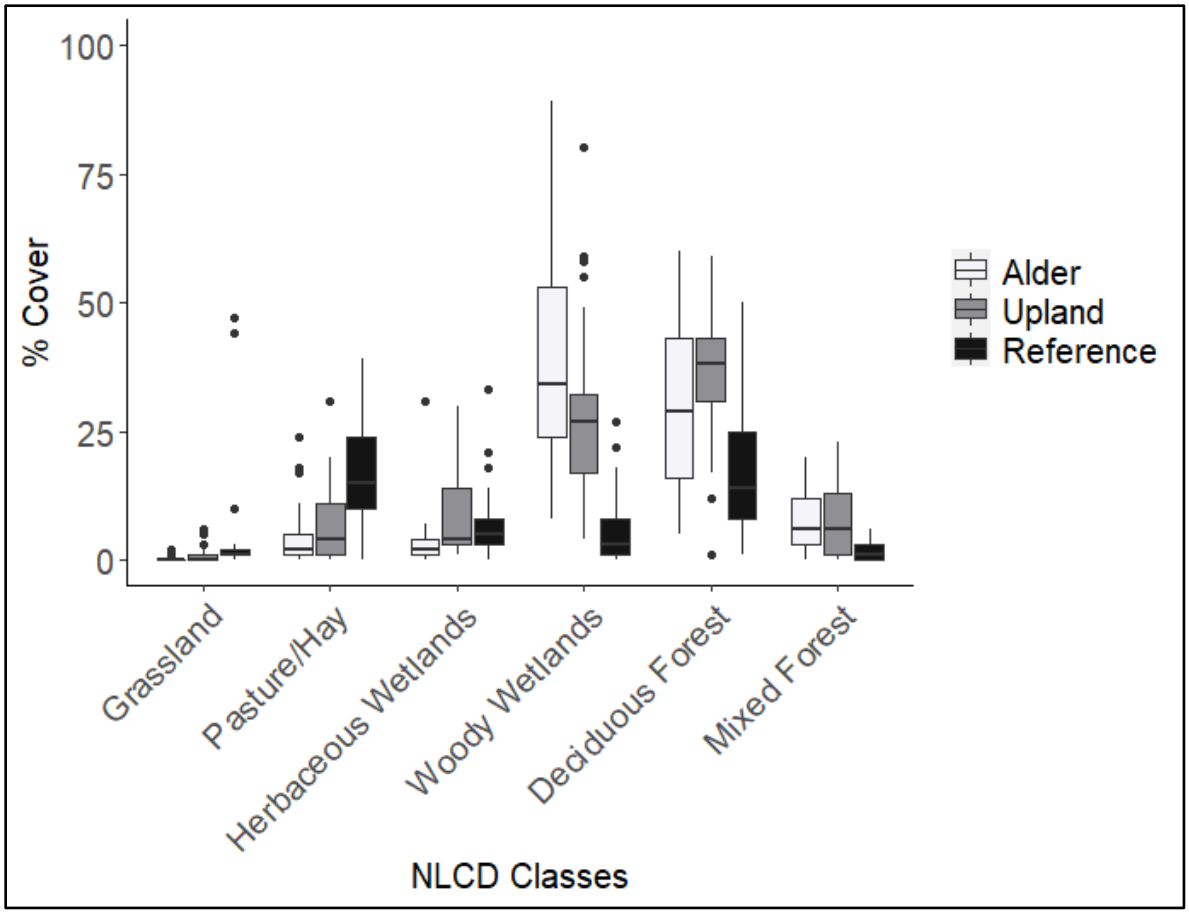

Figure 2. Comparison of percent area of six land cover classes extracted from a 2km radius around the study site for alder (white), upland (gray), and reference sites (black). Study sites are in northern Minnesota and Wisconsin and were monitored three times between June and August 2021. Middle line within the box represents the median, the box itself represents the interquartile range (IQR, 25th – 75th percentile), the vertical lines represent minimum and maximum values (1.5*IQR), and the dots represent outliers.

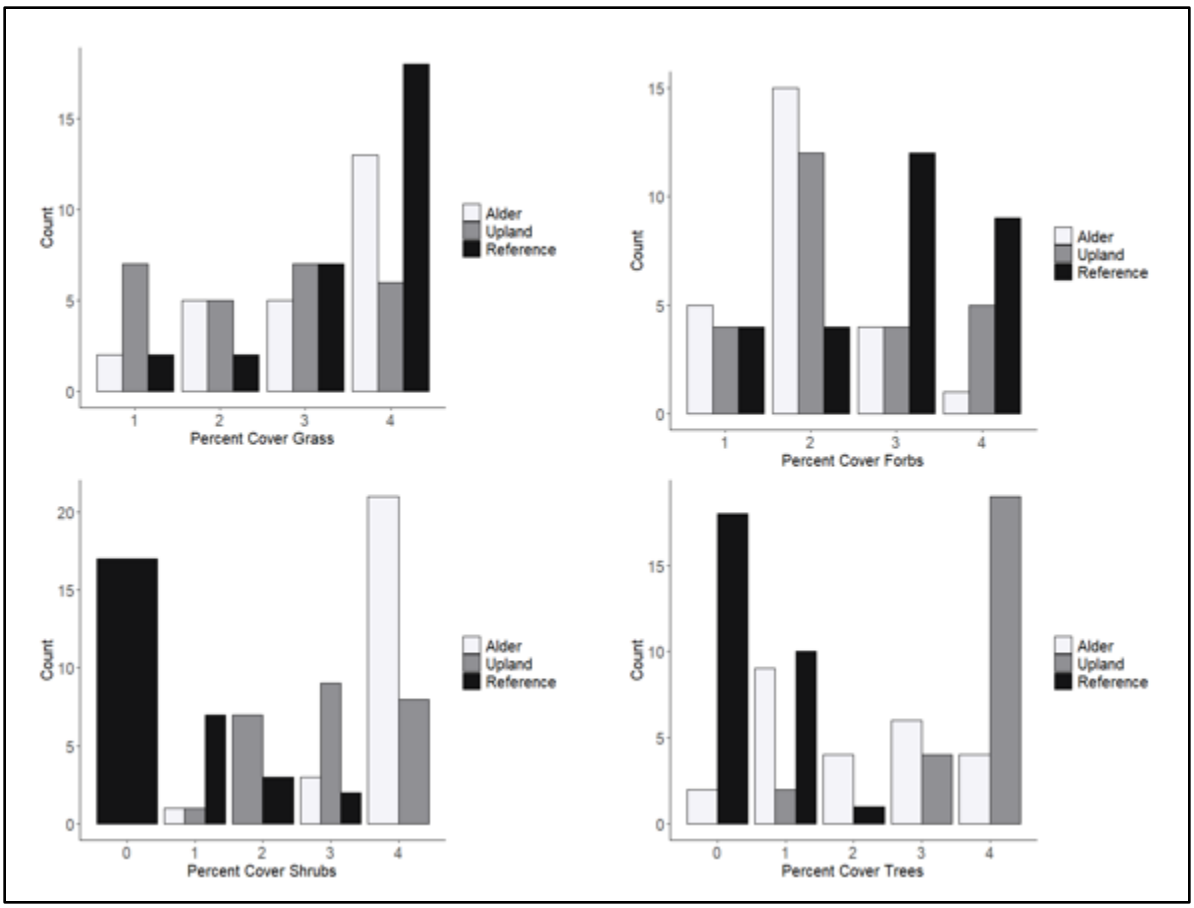

Figure 3. Comparison of four structural vegetation classes (grass – top left, forbs – top right, shrubs – bottom left, trees – bottom right) between alder (white), upland (gray), and reference (black) sites. This survey was conducted once for each study site during the first visit in June. Alder and upland sites were visited in June 2021 and reference sites were visited in June 2020 or 2021. All study sites are in northern Minnesota and Wisconsin. Values on x-axis: 0 = 0%, 1 = 1-10%, 2 = 11-25%, 3 = 26-50%, 4 = >50%.

Table 5. Summary statistics comparing frequency of subplots with blooming nectar plant present (NP subplots) and blooming nectar plant species richness (NP richness) for alder, upland, and reference sites. All sites were in northern Minnesota and Wisconsin. Alder and upland surveys were conducted during three visits June-August 2021. Reference site were surveyed between June-August 2020 and 2021.

	Treatment	n	Mean	SD	Min	Max
NP subplots	Alder	25	0.37	0.15	0.17	0.69
	Upland	25	0.38	0.12	0.2	1
	Reference	25	0.59	0.27	0.09	0.7
NP richness	Alder	25	7.13	3.09	3.67	17.67
	Upland	25	10.4	3.25	5	18.33
	Reference	13	10.04	5.14	1.5	18

Butterfly and Bumble Counts

At the alder and upland sites during the pollinator surveys, we observed a total of 1,599 butterflies from 22 different species or groups. The three most abundant groups were skippers (subfamily: *Pyrginae* or *Hesperiinae*), large fritillaries (*Speyeria spp*.), and satyrs (subfamily: *Satyrinae*). Additionally, we observed a total of 1,068 bumble bees (*Bombus spp*). We observed slightly more pollinators per visit at the upland young forest sites (20.4 ± 20.3 pollinators/visit, min=0, max=118) compared to alder shrub sheared sites (15.2 ± 14.1 pollinators/visit, min=0, max=73). We observed the most butterflies early in the summer (first visit) and we observed the most bumble bees towards the end of summer (third visit).

Modeling Results – Logistic Regression

According to our logistic regression model's different habitat characteristics predicted the presence of monarch adults and immatures (eggs and larvae) and management type (alder or upland) was not a predictor for any of the groups (Table 6). The presence of small shrubs was the only predictor for presence of adult monarchs (briers score = 0.22, AUC = 0.67), whereby sites with more shrubs had more adult monarchs. Site average floral abundance (ln(flowers/m^2) was the most consistent predictor for eggs (briers score = 0.16, AUC = 0.78), larvae (briers score = 0.11, AUC = 0.80), and all monarch stages (briers score = 0.22, AUC = 0.78), whereby sites with higher floral abundances had more eggs, larvae, and all monarch stages. The second most frequent habitat characteristic that was a significant predictor was herbaceous wetland land cover for eggs (briers score = 0.13, AUC = 0.80), all monarch stages (briers score = 0.21, AUC = 0.70), and milkweed (briers score = 0.21, AUC = 0.70) (Table 6), whereby sites with more herbaceous wetland land cover in their surrounding landscapes had more eggs, all monarch stages, and milkweed.

Table 6. Relationships from logistic regression analysis that identifies landscape and site-level characteristics that predict the presence adult monarchs, eggs, larvae, all three monarch stages, and milkweed. Data used for this analysis were collected during the adult monarch survey at alder and upland sites in northern Minnesota and Wisconsin. All study sites were visited three times June-August 2021. The green (+) cells indicate positive predictor of presence, red (-) cells indicate negative predictor of presence, and gray cells indicate those variables were not included in the analysis. Treatment includes the two site types (alder, upland), post treat is the number of growing seasons since management, Avg NP subplots is the site average subplots if a blooming nectar plant present, Avg ln(flowers/m^2) is the site average floral abundance, Avg NP is the site average blooming plant species richness, Avg milkweed density is site average density of milkweed stems/m^2, and milkweed presence is binary (1=present, 0=absent).

		Adult Monarch	Eggs	Larvae	All Monarch Stages	Milkweed
Model set 1	Treatment					
	Post Treat					
Model set 2	Elevation (m)					
	latitude					
	longitude					
	Grassland (71)					
	Pasture/Hay (81)		+			+
	Herbaceous Wetlands (95)		+		+	+
	Deciduous Forest (41)					
	Mixed Forest (43)				-	
	Woody Wetlands (90)					
Model set 3	Avg NP subplots		+			
	Avg ln(flowers/m2)		+	+	+	
	Avg NP richness					
	Avg milkweed density		+			
	Milkweed presence					
Model set 4	canopy cover (%)					
	large sapling cover (%)					
	small sapling cover (%)					
	large shrub cover (%)					
	small shrub cover (%)	+				
	rubus cover (%)					
	forb cover (%)					
	fern cover (%)					
	grass/sedge cover (%)					
	cwd cover (%)					
	leaf litter cover (%)					
	ground cover (%)					
	stand basal area (m^2/ha)					

Modeling Results – HDM across season

Our models detected no difference in butterfly or bumblebee densities between the two community types: alder and upland sites. Throughout the summer, our models predicted butterfly density of 101.5/ha (lower=84.6, upper=121.8, CI=18.5) and bumble bee density of 83.9/ha (lower=66.4, upper=106.1, CI=19.7). For the butterfly analysis, there was no significant variables at the site level analysis, but at the landscape level, elevation was a significant negative predictor of butterfly density ($QAIC_c$ weight = 0.39, β 95% CI = -0.06, -0.4) (Figure 4). For the bumble bee analysis, the frequency of subplots with blooming nectar plants was a positive predictor ($QAIC_c$ weight = 0.18, β 95% CI = 0.8, 0.5) (Figure 4) at the site level, and there were no significant predictors at the landscape level.

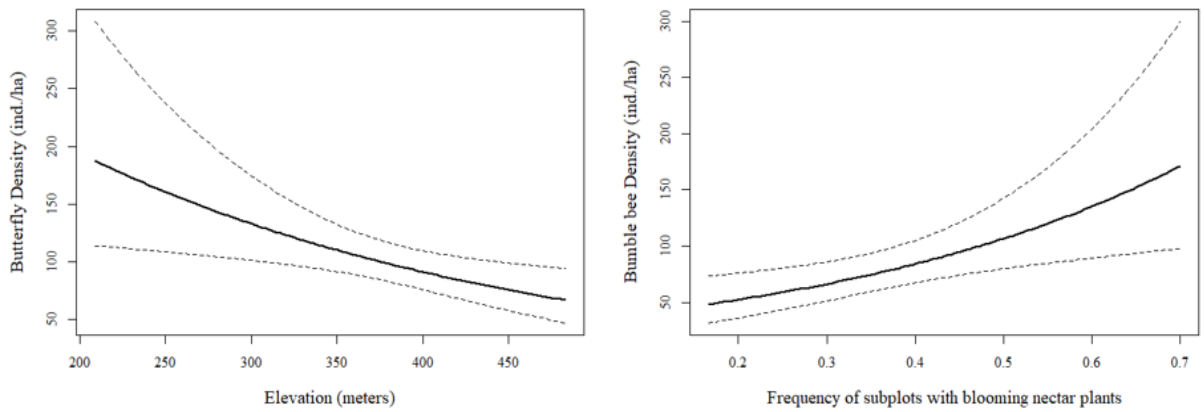

Figure 4. Model results for the butterfly and bumble bee HDM across season analysis. For butterfly and bumble bee data we ran two groups of models independently (landscape and local scale). These models are comparing butterfly and bumble bees that were recorded at alder and upland sites in northern Minnesota and Wisconsin and sampled during three rounds, June-August 2020. For the landscape analysis, elevation was a negative predictor for butterfly density (left graph) and for the site level analysis, frequency of subplots with blooming nectar plants was a positive predictor for bumble bees (right graph). The dark middle line shows the model prediction, and the dashed lines are 95% confidence intervals.

Modeling Results – HDM within season

Models did not detect a difference in butterfly or bumble bee densities between alder and upland sites during each of the three survey rounds. These models predicted butterfly densities to be highest during the first survey round (June) (49.6 butterflies/ha, lower=43.9, upper=55.9, CI=5.9) and lowest during the third survey round (August) (17.4 butterflies/ha, lower=2.8, upper=12.7, CI=5.5). One explanatory variable predicted butterfly density; stand basal area during the first survey round and it was positive (Figure 5). Bumble bee density was predicted by multiple explanatory variables (Fig. 5 and 6). During visit one (June), the percent cover of brambles was a positive predictor of bumble bee density, during visits two (July) and three (August) floral species richness and frequency of subplots with a blooming nectar plant present were positive predictors of bumble bee density (Figure 5 and 6). Additionally, during visit three, large and small shrub cover were negative predictors of bumble bee density (Figure 6). Our models predicted bumble bee densities to increase throughout the summer (Visit 1: 40.1 bumble

bees/ha, lower=16.6, upper=99.7, CI=36.5; Visit 3: 90.6 bumble bees/ha, lower=66.5, upper=123.6, CI=28.1).

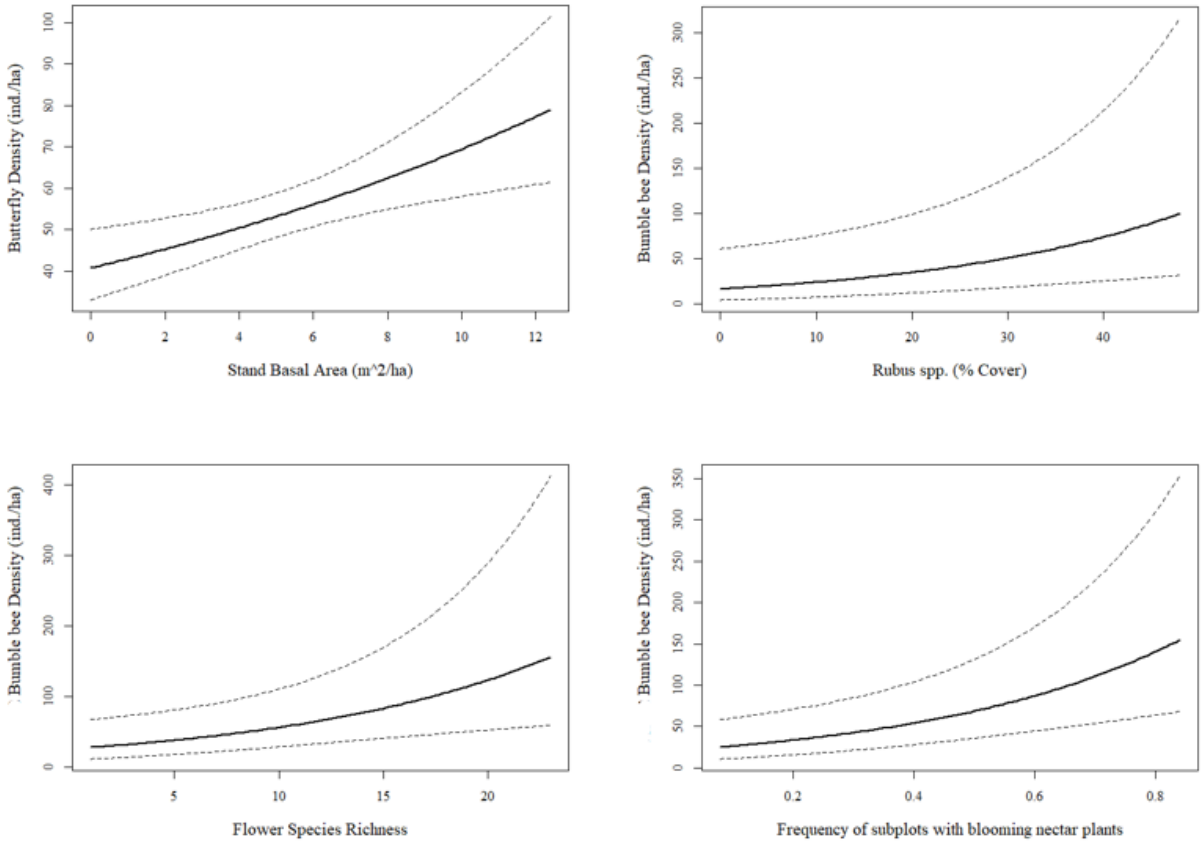

Figure 5. Model results for the butterfly and bumble bee HDM within season analysis. These models are comparing butterfly and bumble bees that were recorded at alder and upland sites in northern Minnesota and Wisconsin and sampled during three rounds, June-August 2020. Top left, stand basal area is a positive predictor for butterfly density during visit one (June) ($QAIC_c$ weight = 0.90, β 95% CI = 0.3, 0.1). Top right, bramble (Rubus spp.) cover is a positive predictor for bumble bee density during visit one (June) ($QAIC_c$ weight = 0.39, β 95% CI = 0.9, 0.1). Bottom left, flower species richness is a positive predictor for bumble bee density during visit two (July) ($QAIC_c$ weight = 0.66, β 95% CI = 0.7, 0.2), as well as frequency of subplots with blooming nectar plants (bottom right) ($QAIC_c$ weight = 0.13, β 95% CI = 0.6, 0.1). The dark middle line shows the model prediction, and the dashed lines are 95% confidence intervals.

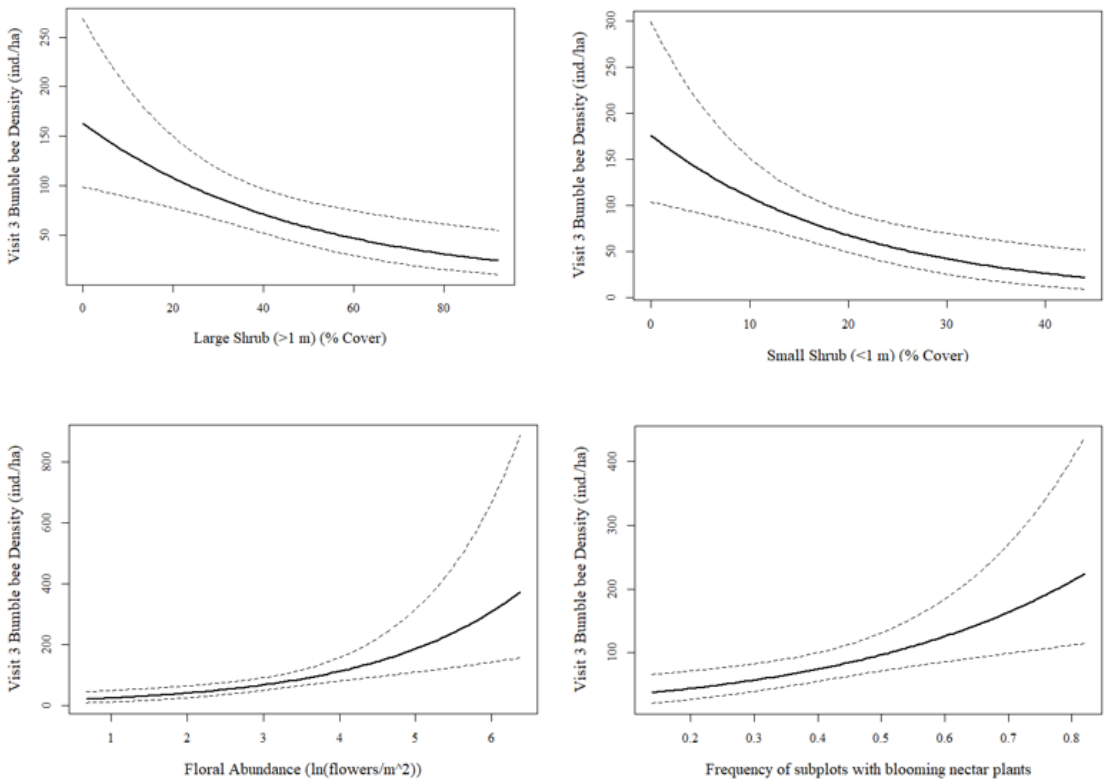

Figure 6. Model results for bumble bee HDM within season analysis during visit three (August). All bumble bee observations were recorded at alder and upland sites in northern Minnesota and Wisconsin and sampled during three rounds, June-August 2020. These four graphs show the relationships of all significant variables that predicted bumble bee density during visit three (August), both large and small shrub cover are negative predictors and both floral abundance and frequency of subplots with blooming nectar plants are positive predictors. Top left, large shrub % cover ($QAIC_c$ weight = 0.10, β 95% CI = -0.2, -0.8). Top right, small shrub % cover ($QAIC_c$ weight = 0.11, β 95% CI = -0.2, -0.7). Bottom left, floral abundance (ln(flowers/m^2) ($QAIC_c$ weight = 0.62, β 95% CI = 0.8, 0.2). Bottom right, frequency of subplots with a blooming nectar plant present ($QAIC_c$ weight = 0.14, β 95% CI = 0.8, 0.2). The dark middle line shows the model prediction, and the dashed lines are 95% confidence intervals.

Future activities

The overarching goal of this research was to examine monarch butterfly use of managed alder and upland (young forest) communities, and to examine the influence of vegetation and landscape characteristics on monarch presence. Our preliminary results suggest that upland and alder sites 1-5 years post-management hosted similar densities of adult and immature monarchs, albeit lower densities than MJV reference sites. Future activities for this project include running a final logistic regression model to compare presence of monarchs and milkweed among alder, upland, and reference sites. From these models we will provide a final evaluation of what landscape and local habitat characteristics are important predictors for monarchs and milkweed. Finally, the graduate student will present our results at two meetings:1) Midwest Association of Fish and Wildlife Agencies annual conference and 2) the North Central Branch Entomological Society of America. We also intend to submit two manuscripts for publication by late 2022.

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Part II: A Multi-Regional Assessment of Eastern Whip-poor-will (*Antrostomus vociferus*) Occupancy Within Private Forests Enrolled in NRCS Conservation programs that target Golden-winged Warbler (*Vermivora chrysoptera*), Cerulean Warbler (*Setophaga cerulea*), and New England Cottontail (*Sylvilaegus transitionalis*)

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Abstract

The Eastern Whip-poor-will has experienced significant population declines over the past half century. One factor contributing to this decline, and that of many other wildlife species, is the loss of young forest conditions. Several agencies and conservation organizations have initiated efforts to implement management practices that address this habitat deficiency. The NRCS's Working Lands for Wildlife and Regional Conservation Partnership Programs are two such efforts that target the creation of young forest conditions for Golden-winged Warblers and New England Cottontails (WLFW) and structurally complex variable density older forests for Cerulean Warblers (RCPP). Understanding how other at-risk species, like the whip-poor-will, respond to these efforts is necessary to quantify potential benefits of these conservation programs. We implemented an automated recording unit ("ARU") based monitoring protocol to overcome the logistical challenges of assessing whip-poor-will occupancy across a large geographic extent within a single field season. In 2020 and 2021, we deployed 630 ARUs across hundreds of private sites enrolled in WLFW and RCPP ranging from western North Carolina to southern Maine. Twenty-eight (4%) of our units failed due to bear or water damage. Over 20,000 hours of nocturnal recordings were collected across all functioning ARUs. We developed a machine learned classifier to detect whip-poor-will song. We used classifier results to create detection histories for each survey location and then modeled whip-poor-will occupancy while accounting for imperfect detection. Whip-poor-will were detected at 264 of 602 survey locations (naïve occupancy = 43.8%). Preliminary results indicate whip-poor-will detection was most influenced by total minutes of recordings (+). Modeled occupancy probability was 42%. Whip-poor-will occupancy was most influenced by woody stem density (+) at the site level and nine (n=9) landscape level covariates. Our project demonstrates the value of using ARUs to implement a large-scale monitoring program to evaluate conservation outcomes. Future efforts involve processing recordings collected in 2021 at and additional >1100 sites on public forests and rerunning all preliminary analyses described herein.

Introduction

North American birds are experiencing drastic population declines with an estimated loss of nearly three billion individuals over the past half-century (1970-2017; Rosenberg et al. 2019). Eastern forest birds and aerial insectivores are among the most declining groups of birds with an estimated population reduction of nearly 170 million individuals (15-19%) and 160 million individuals (26-36%), respectively (Rosenberg et al. 2019). It is not surprising that species belonging to both groups are some of the most imperiled. For example, the Eastern Whip-poor-will (*Antrostomus vociferus*; hereafter whip-poor-will), a nocturnal aerial insectivore that

inhabits eastern forests from Northern Georgia to Southern Ontario and west to Minnesota and Oklahoma (Cink et al. 2020), has experienced a 64% population reduction (1.9 % loss/year) from 1966-2017 (Pardieck 2020). Possible factors contributing to whip-poor-will population declines include habitat loss and degradation, as well as reduced food availability related to pesticides and climate change (Spiller and Dettmers 2019).

Whip-poor-will use a variety of forest conditions and landscape contexts during the breeding season (Cink 2020, Vala et al. 2020, Wilson and Watts 2008). The species relies on early successional communities for foraging, but there is mixed evidence suggesting they also need mature forests or forest ecotones for diurnal roosting and nesting (Wilson and Watts 2008, Akresh and King 2016). Forest management is known to be an important tool for creating and maintaining local landscape conditions that are attractive to whip-poor-will (Wilson and Watts 2008, Spiller and King 2016, Farrell et al. 2017). A study within forests dominated by pine dominated forests in the southern portion of the species range recommended that forest management that incorporates practices that create and maintain regenerating forest patches proximate to mature forests should benefit whip-poor-will (Wilson and Watts 2008). Similarly, studies conducted in the northern portion of the species range found that sites proximate to young forest patches and shrublands had higher occupancy than mature forest dominated sites (Tozer et al. 2014, Grahame et al. 2021). Additionally, whip-poor-will occupancy was positively correlated with forest patch size and amount of wetland cover and was unaffected by the type of cropland present in the landscape (hay/pasture vs cropland; Vala et al. 2020). More research is needed across the species breeding range and in relation to different forest management practices to fully understand the species ecology (Akresh and King 2016). Moreover, a large portion of the whip-poor-will's breeding distribution falls within landscapes heavily dominated by eastern deciduous forests (Cink 2020), yet no studies have investigated factors that influence broad scale patterns of whip-poor-will distribution within this forest type and associated landscape contexts.

Deciduous forests of the eastern US are dominated by uniformly-aged older forests (Shifley et al. 2014). As such, many species that require more diverse forest conditions are in decline (Fiss et al. 2020, King and Schlossberg et al 2014, Litvaitis 2001). Several conservation efforts are underway on public and private lands to address the need for creating and sustaining diverse forest conditions that promote the recovery of many declining forest wildlife (Bauer 2018, Nareff et al 2019, Litvaitis et al. 2021). While monitoring of species that are the target of these conservation efforts has been completed (i.e., Golden-winged Warbler *(Vermivora chrysoptera),* McNeil et al. 2020, New England Cottontail (*Sylvilagus transitionalis;* Bauer 2018) and Cerulean Warbler (*Setophaga cerulea*, Nareff et al. 2019), an assessment of how whip-poor-will respond to these forest management efforts has not occurred. Given the species' imperiled conservation status, it is valuable to understand the extent to which forest management that targets other species may benefit whip-poor-will and to identify which management practices and landscape contexts result in high occupancy. Additionally, little is known about whip-poor-will requirements and thresholds regarding forest patch size, residual basal area, and time since management within landscapes dominated by eastern deciduous forest.

The very specific nocturnal and lunar phase survey protocol for whip-poor-will have limited researchers' ability to evaluate the potential benefits of forest management across large geographic extents. Most surveys for whip-poor-will are conducted from roads (i.e. Nightjar Survey Network), thus limiting inferences that can be made to the species response to stand-level forest management that often occurs well away from accessible roads (Betts et al. 2007). The recent availability of low-cost autonomous recording units (ARUs) can now facilitate off-road nocturnal surveys across hundreds of managed forests in a single breeding season (Hill et al. 2019). ARUs also provide researchers the ability to survey whip-poor-will at each site every night throughout the entire breeding season. The deployment of hundreds of ARUs across managed landscapes has significant potential to increase our understanding of whip-poor-will habitat associations, regional population trends, behavior, and response to forest management.

In this study, our goal is to develop and implement an ARU-based regional monitoring protocol to assess whip-poor-will occupancy across various landscapes contexts, silvicultural treatments, and forest types in the Appalachian Mountain and New England regions. Our objectives are to: 1) assess whip-poor-will response to forest management associated with United States Department of Agriculture's (USDA) Natural Resource Conservation Service's (NRCS) Working Lands for Wildlife and Regional Conservation Partnership programs; 2) understand the distribution of whip-poor-will in portions of the Appalachian Mountains and New England regions; and 3) identify within stand and landscape level factors that influence whip-poor-will site occupancy and range wide distribution. This research will improve our understanding of whip-poor-will ecology and also serve as a case study for implementing ARU-based protocol across a large spatial extent that would not be logistically feasible using human observers. Lastly, my research will produce habitat- and region-specific management recommendations for whip-poor-will and will help fulfill the USDA-NRCS's goal to assist private landowners in creating high quality habitat for threatened and declining wildlife.

Methods

Study Area

Survey locations ranged from western North Carolina to southern Maine. This geography represents a latitudinal extent that spans from the core to the periphery of the whip-poor-will's breeding range in the eastern US. Sites primarily fell within Eastern Temperate Forest ecoregion; however, some sites were within the Northern Forests ecoregion (US EPA 2013). Tree species composition varied across survey locations; however, sites were predominantly comprised of deciduous tree species (e.g. *Quercus* spp., *Acer* spp., and *Carya* spp.). Elevation at sites ranged between 6 and 1199 m above sea level.

Study Design

In 2020 and 2021, we deployed ARUs across hundreds of managed public and private forests. We deployed 129 ARUs from April-July 2020 across private lands in Pennsylvania and Maryland enrolled in NRCS's Regional Conservation Partnership Program to benefit Cerulean Warbler (CERW; *Setophaga cerulea*; n= 129 sites). Additionally, we deployed 501 ARUs from April-July 2021 at sites enrolled in the NRCS's Working Lands For Wildlife to create habitat for the Golden-winged Warbler (GWWA; n= 311 sites) and New England Cottontail (NEC; *Sylvilagus transitionalis;* n=190 sites)(Fig.1; Table 1). Thus, across both years, we deployed ARUs at a total of 630 locations in treatments that ranged from clearcuts and overstory removals with residuals to first and second entry shelterwoods (Photo Set 1).

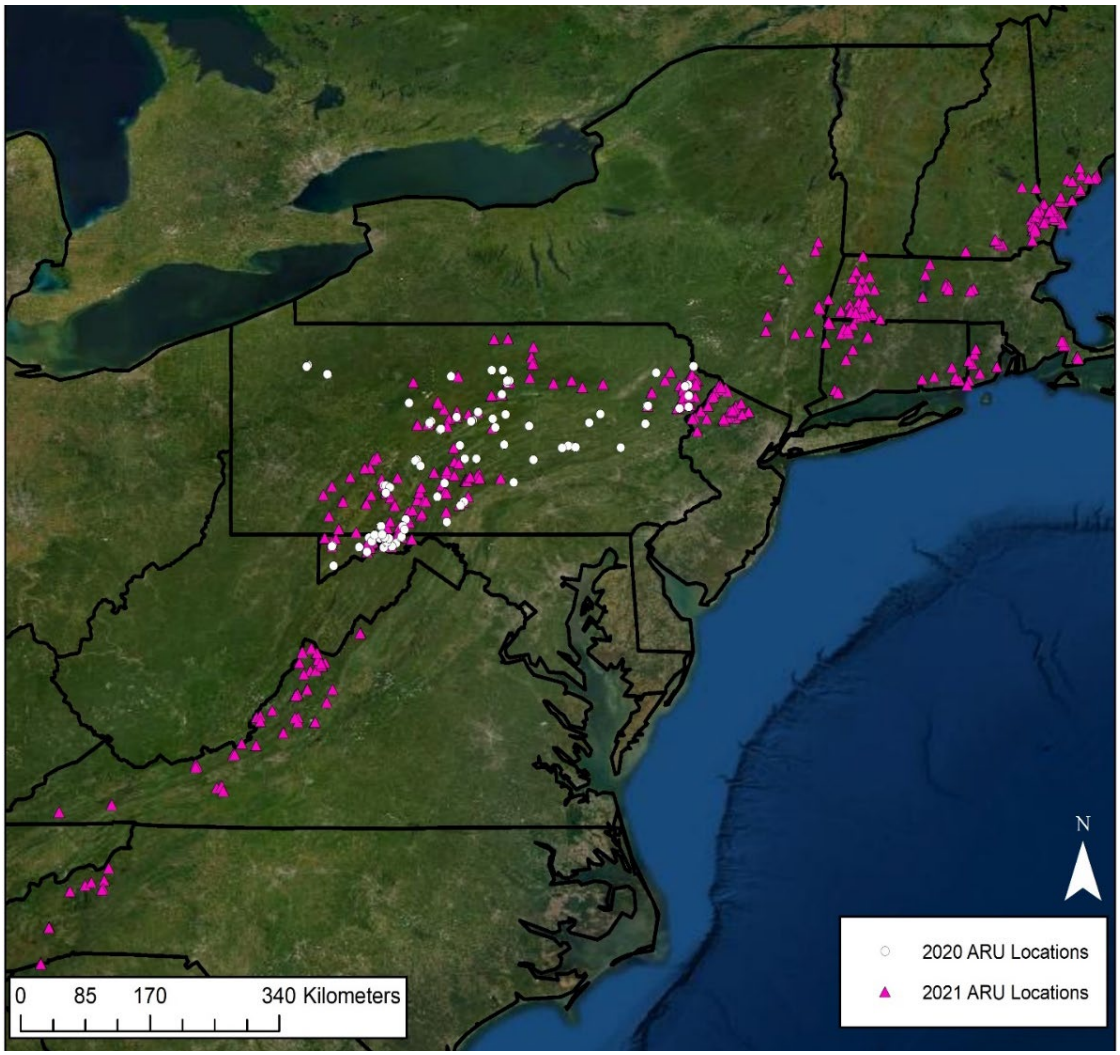

Figure 1. Distribution of study sites throughout the eastern United States where managed forests were surveyed for Eastern Whip-poor-will. Three types of managed forests (clear cuts, overstory removals, and shelterwoods) were surveyed from April-July 2020 and 2021. Note: locations of private sites are randomly shifted to protect landowner privacy.

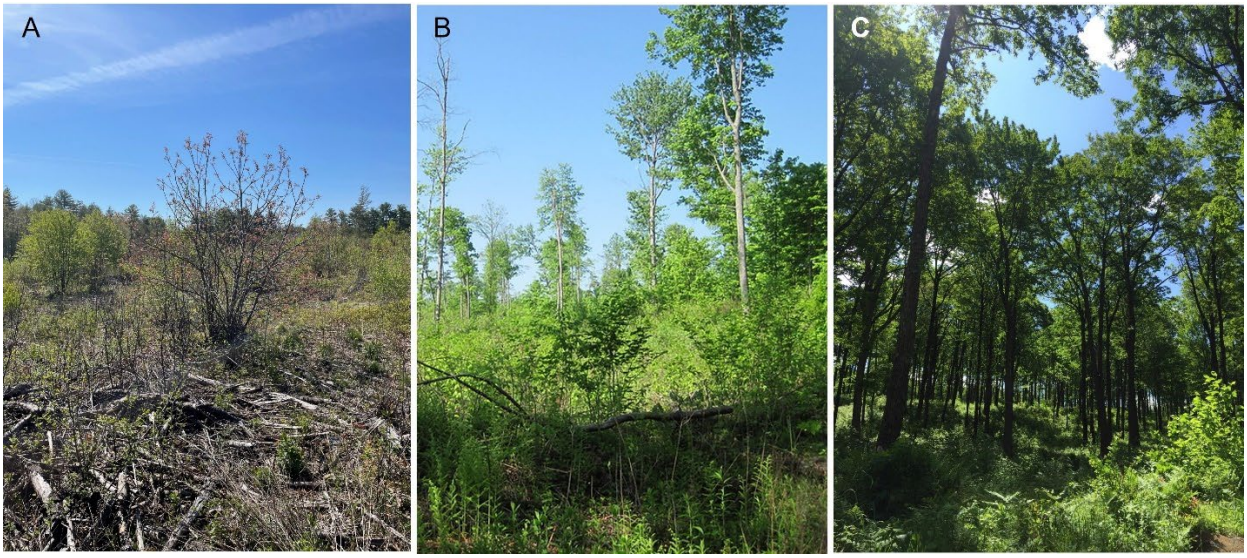

Photo set 1: Representative photos of the managed forests communities from western North Carolina to southern Maine where we surveyed for Eastern Whip-poor-wills. Specifically, we investigated three types of managed forest communities from April-July 2020 and 2021; A) clearcuts B) overstory removals with residuals, and C) shelterwood harvests.

Using ArcGIS 10.6.1, we randomly plotted points within managed and unmanaged stands on each property. Unmanaged sites had points placed within the planned management boundary. ARUs were at least 50 m from the harvest edge to reduce edge effects and a minimum of 500 m from another survey location to reduce the likelihood of double counting individuals (Ruff et al. 2020, Nightjar Survey Network).

Table 1. Number of Autonomous Recording Units (ARUs) deployed for Eastern Whip-poor-will by year, project, and state.

Year	Project	State	Number of Sites
2020	CEAP CERW	PA	94
		MD	35
		Total	**129**
2021	WLFW GWWA	NC	13
		VA	52
		MD	33
		PA	165
		NJ	48
	WLFW NEC	NY	12
		CT	26
		RI	9
		MA	60
		NH	44
		ME	39
		Total	**501**

Autonomous Recording Units

We used ARUs (Audiomoths, developed by Open Acoustic Devices) to collect audio data at each survey location. ARUs were deployed to capture at least one of the survey windows in 2020 which are from April 30th to May 14th and May 29th to June 13th (Nightjar Survey Network). In 2021, units were deployed from May 18th to July 2nd. These dates encapsulated the 2021 nightjar survey windows of May 19th-June 2nd and June 17th-July 1st (Nightjar Survey Network). We used the ARU setup and deployment protocol developed by our collaborators the University of Pittsburgh's Bioacoustics Laboratory (Rhinehart 2019). ARUs were updated to firmware version 1.5.0 and equipped with a 64 gigabyte SD card. Additionally, each ARU was programed to record for 2 hours after sunset and 15 minutes before sunrise to capture whip-poor-will songs. We placed each ARU in a quart-sized Ziploc freezer bag containing silica desiccant to absorb potential moisture. A ziptie was used to attach each ARU to a sapling (>3 cm in diameter) or tree closest to the randomly generated survey location at a height of 1.5-2 m from the ground (Photo set 2). All ARUs were recovered in early July after the second survey period.

Photo set 2: Photos of the autonomous recording units used to survey Eastern Whip-poor-wills in managed forest communities from western North Carolina to southern Maine from April-July 2020 and 2021.

Vegetation Sampling

To quantify structural vegetation around each ARU location, we conducted a single bi-radial vegetation survey from June 1-July 15, 2020 and 2021 at all ARU locations (Fig. 2). Specifically, a 35-m radial transect was oriented in 2 of the following directions, 0°, 120°, and 240°, radiating from point center (Figure 1). At each plot's center and the end of each transect, we used a 10-factor wedge prism to estimate basal area, a point quarter plot, and a 1 m^2 ground cover plot (n = 3 total readings/point). The point quarter plots were be divided into 4 quadrants: North-East, East-South, South-West, and West-North. In each quadrant, the species of closet tree (>10 cm diameter at breast height, DBH) to plot center was recorded in addition to its distance and DBH. Trees further than 15 m from plot center were considered out of the plot. A 1m^2 ground cover plot was used to estimate the percent woody stems (<0.5 m), herbaceous cover, leaf litter, fern, bare ground/rock, *Rubus* spp., and moss. Each variable was be rounded to the nearest whole percent and estimated. On each 35-m transect all woody stems <10 cm DBH were counted within a 1-m wide transect from 15-25 m from point center. Stems were be identified to species and grouped into 2 categories: small (0.5-1.5 m in height) and large (>1.5 m in height). Along the entire length of each 35 m transect I recorded the number of downed woody material pieces (> 10 cm DBH) crossed.

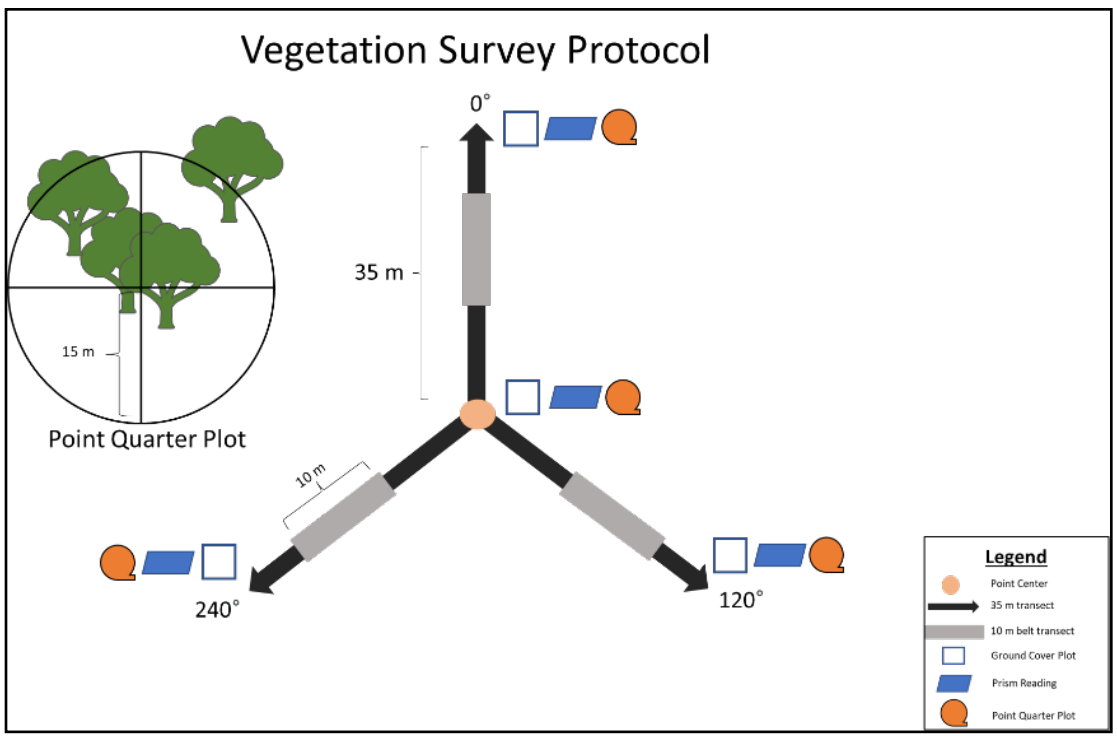

Figure 2. Layout of the vegetation surveys, including locations of 10 m transects, ground cover plots, point quarter plots, and prism readings. Observers will randomly select 2 of the 3 transect directions, thus the number of measurements will be one less than those shown. The vegetation survey was completed within managed forest communities surveyed for Eastern Whip-poor-wills ranging from western North Carolina to southern Maine from April-July 2020 and 2021.

Landscape Assessment

We used the "summarize categorical raster" tool in ArcGIS Pro 2.9.1 to characterize the local landscapes surrounding each ARU location. We used the National Land Cover Dataset (NLCD 2019; Dewitz 2019) to quantify the percent composition of forest, wetland, agricultural, human development, and other land cover types in the landscape within 500 m of the survey location. We also used the NRCS's "USA SSURGO - Drainage Class" dataset from ArcGIS online to calculate the percent of poorly, well, and excessively drained soils within 500 m (USDA NRCS 2017). We used the US Forest Service's Forest Inventory and Analysis dataset (USDA Forest Service 2019) to determine the forest community types around each survey location, and the United States Geological Survey's Global Multi-resolution Terrain Elevation Data (GMTED 2010; Danielson and Gesch 2011) to calculate the elevation at each survey location. Using the NRCS's soil drainage dataset, we quantified the percent of poor, well, and excessively drained soils within 500m of the point. Future steps include, using imagery from the National Agricultural Imagery Program (NAIP 2019) to digitize management boundaries to determine patch size and measure distance to intact forest edge. Additionally, we will use the National Cooperative Soil Survey (NCSS) dataset to determine soil type(s). All landscape variables will be quantified for several spatial extents (i.e. 500 m, 1 km, 2 km, and 5 km; Vala et al. 2020).

Acoustical Processing

After the ARUs were collected from the field, we downloaded the recoding data to an external hard drive and cloud-based server. We then split recordings into 5-second clips (e.g., 1 hour of recording = 720, 5-second clips) and ran them through an automatic whip-poor-will classifier developed by our collaborators at the University of Pittsburgh. The classifier assessed each 5-second clip for the presence of whip-poor-will song and assigned each clip a score. The classifier score is the likelihood that a clip contains a whip-poor-will song. The higher the classifier score, the more likely the clip is to have a recording of a whip-por-will song.

We used a top-down listening approach to assesses the presence of whip-poor-will at each ARU. Top-down listening is an approach where a score threshold is selected and clips scoring higher than the selected score is assessed. We divided the 2 survey windows into 4 listening bouts so we could model for imperfect detection (B1 = May 19th - May 25th, B2 = May 26th - June 2nd, B3 = June 17th - June 24th, B4 = June 25th - July 1st). First, we selected a threshold of 10 and assessed all site-window combinations (i.e. ARU-0001, B1; ARU-0001, B2; ARU-0001, B3; ARU-0001, B4) that had files with scores higher than 10. All site-window combinations where whip-poor-will were detected were removed from further listening as that combination was considered occupied. We repeated this process for thresholds of 8,7, 6, and 5 until we found the score where false positives began (score = 6.67). The results of this report are from listing down to a threshold of 5 with our current classifier, but in future analyses we will start our listing at a threshold of 6.68. Additionally, we are working with our collaborators to create a second-generation whip-poor-will classifier which will further increase detection and thus likely set a new threshold score. Regardless, the results presented herein should be considered minimum occupancy.

Occupancy modeling

We modeled whip-poor-will presence using static occupancy models in the R package *unmarked* (Fiske & Chandler 2011; R Core Team 2020). Package unmarked allowed us to fit linear models within a maximum likelihood framework that can be combined with an information theoretic approach (Andersen 2007) for model selection (e.g. using Akaike's information criterion adjusted for small sample size; AICc; Burnham & Andersen 2002). We accounted for detection probability by modeling factors that influence detection probability using three survey covariates: (1) Julian date, (2) Minutes of recording, (3) stem density. We created all possible combinations of 0–3 survey covariates on detection probability (p; Barton 2018; R Core Team 2020). We considered covariates to be informative if they were in the competing set (<2.0; ΔAICc) and had β95% confidence intervals (CIs) that did not include zero (Burnham & Andersen 2002). After establishing informative survey covariates on detection probability, we incorporated them into consecutive occupancy models assessing habitat patterns. We included 2 site level (1-2) and 12 landscape level covariates (3-14) in our occupancy models: (1) stem density, (2) basal area, (3) elevation, (4) percent well drained soil, (5) percent water, (6) percent high intensity development, (7) percent barren land, (8) percent forest cover, (9) percent shrub/scrub cover, (10) percent herbaceous cover, (11) percent row crop cover, (12) percent oak/pine forest, (13) percent oak/hickory forest, and (14) percent elm/ash forest. We used the function dredge here as well to create all possible combinations of site and landscape level covariates that influenced occupancy. Additionally, we considered covariates to be informative if they were in the competing set (<2.0; ΔAICc) and had β95% confidence intervals (CIs) that did not include zero (Burnham & Andersen 2002). Prior to analysis, we removed all covariates that were strongly correlated, r > 0.6 (Sokal and Rohlf 1969). Also, we calculated AUC using the *dismo* package (Hijmans et al. 2017, R Core Team 2020). AUC is used to determine how well the model fit our data and returns a value between 0-1 (closer to 0 = better fitting model). Finally, we calculated Briers score which is used to assess the fit of a model.

Results

In 2020, we deployed units across 129 sites (Table 1). Five of the 129 units failed to collect data. In 2021, we deployed 501 units in managed forests ranging from western North Carolina to southern Maine (Table 1), of which 23 units failed. We detected whip-poor-will at 264 of the 602 (n= 72 sites in 2020 and n=192 sites in 2021) sites for which we had recordings (naïve occupancy of 44%).

Our occupancy model revealed that the top detection only model contained the covariate "minutes of recording", whereby as minutes of recording increased, detection increased. Our top site level only model contained stem density (briers score = 0.24, AUC = 0.60). Whereby, occupancy increased with higher stem densities (Fig. 3). The top landscape level model contained the covariates percent barren land, high intensity development, forest, shrub/scrub, and water, percent elm/ash, oak/hickory, and oak/pine forest types, and elevation. Whereby, whip-poor-will occupancy decreased with percent of high intensity development and elevation, but

increased with greater percent barren land, forest, shrub/scrub, and water cover types, and percent elm/ash, oak/hickory, and oak/pine forest types (Fig. 4) (briers score = 0.21, AUC = 0.72). Our top model estimated that the average probability of a site being occupied was 42% (LL 37% - UL 47%).

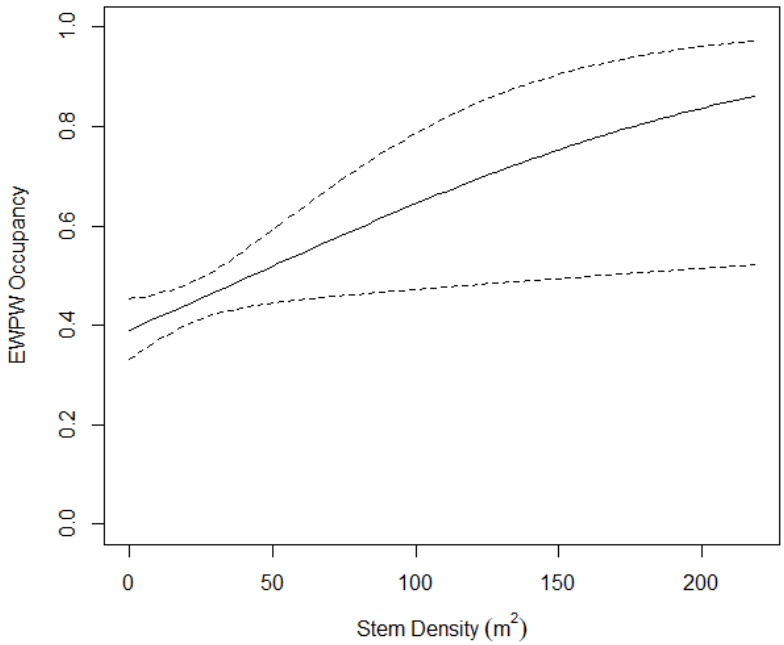

Figure 3. Graph showing relationship between woody stem density and Eastern Whip-poor-will occupancy in managed forest communities from western North Carolina to southern Maine. Surveys were conducted between April-July 2020 and 2021.

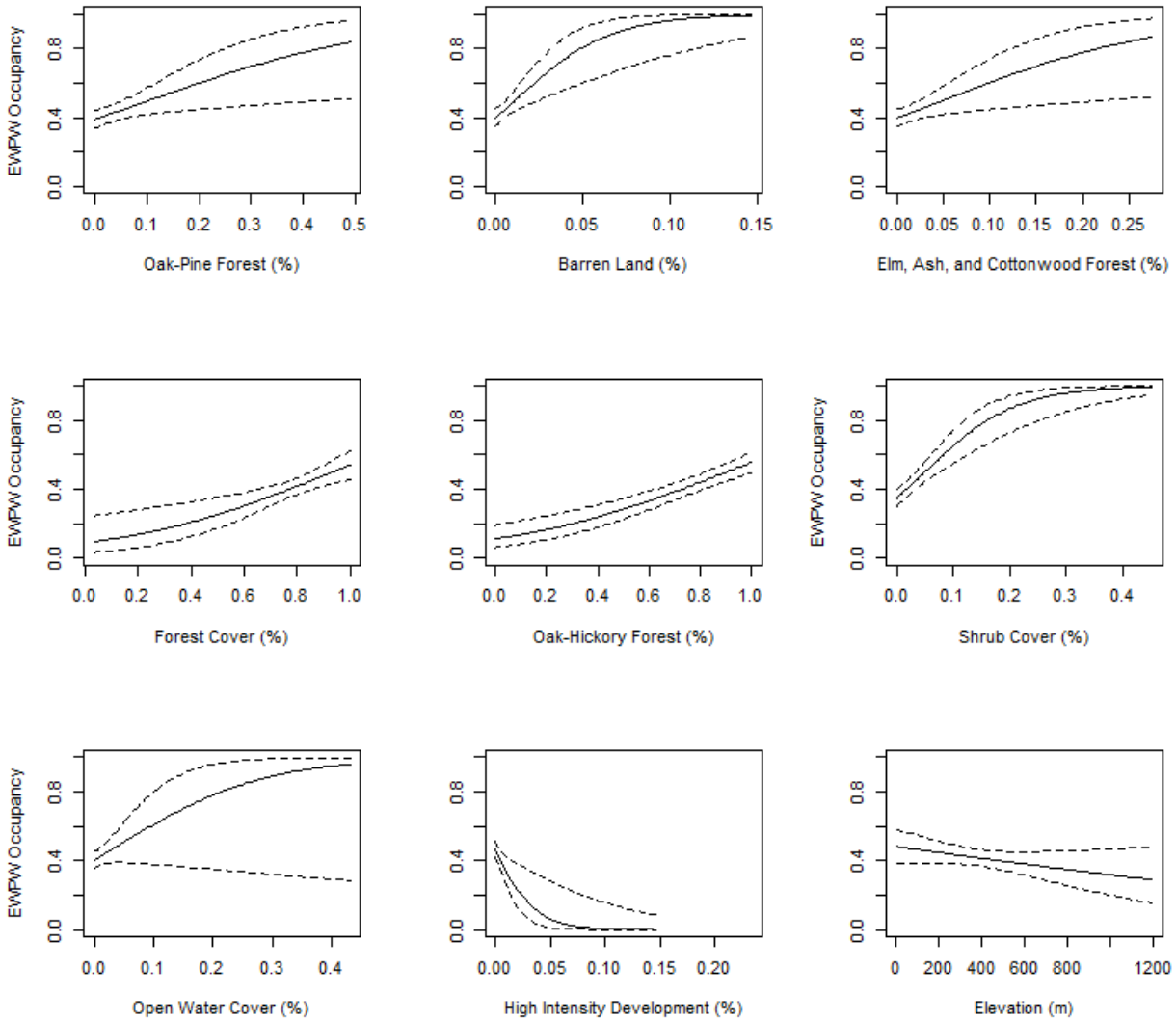

Figure 4. Graphs showing relationships between nine landscape features that were found to most influence Eastern Whip-poor-will occupancy in managed forest communities from western North Carolina to southern Maine. Surveys were conducted between April-July 2020 and 2021.

Table 2. Summary statistics of habitat covariates found to influence Eastern Whip-poor-will occupancy at managed forest sites from western North Carolina to southern Maine. Sites were surveyed using autonomous recording units during April-July 2021 and 2022.

Variable	Unit	N	Mean	SD	Median	Min	Max
Stem Density	10 m^2/ha	589	22.05	19.97	17.50	0.00	218.50
Elevation	m	602	387.89	232.74	375.50	6.00	1199.00
Open Water	% w/in 500m	602	0.01	0.03	0.00	0.00	0.43
High Intensity Development	% w/in 500m	602	0.01	0.02	0.00	0.00	0.23
Barren Land	% w/in 500m	602	0.00	0.01	0.00	0.00	0.15
Forest Cover	% w/in 500m	602	0.80	0.20	0.86	0.04	1.00
Scrub Shrub Cover	% w/in 500m	602	0.02	0.05	0.00	0.00	0.45
Oak-Pine Forest	% w/in 500m	602	0.03	0.07	0.00	0.00	0.49
Oak-Hickory Forest	% w/in 500m	602	0.76	0.35	0.99	0.00	1.00
Elm-Ash Forest	% w/in 500m	602	0.01	0.04	0.00	0.00	0.27

Future activities

We plan to have the second-generation classifier created and ran on recordings from our 602 private land locations presented here and from an additional 1,100 sites on nearby public lands. Landscape data processing will be completed by early April. Additionally, we have completed diurnal point counts for all 478 WLFW survey locations and plan to share results with landowners by the end of spring 2022. Moreover, ARUs recordings can be processed to detect sounds made by other wildlife such as owls, American woodcock, songbirds, wild turkey, and ruffed grouse. Finally, several presentations on this project will be delivered in the coming months, these include Society of American Foresters: Allegheny Plateau Conference, NEAFWA Annual Conference 2022 (planned, abstract not accepted yet), and a public presentation for the New Hampshire Audubon Society. A manuscript on this project will be published by early 2023.

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