

2018 Annual Report

## **Habitat Ecology of Native Pollinators and Imperiled Migratory Songbirds within Early-successional Deciduous Forests**

A Conservation Effects Assessment Project (CEAP)  
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## **Introduction**

The Golden-winged Warbler (*Vermivora chrysoptera*) is one of the most critically threatened, non-federally listed vertebrates in eastern North America. The implementation of science-based best management practices that create or maintain Golden-winged Warbler breeding habitat is thought to be an important step to reversing the species decline. In 2012, NRCS initiated a conservation effort called *Working Lands for Wildlife* (WLFW). This program specifically targets the creation or enhancement of habitat for imperiled species, including the Appalachian population of the Golden-winged Warbler. Additionally, the American Bird Conservancy and its partners were awarded funding for a Regional Conservation Partnership Program (RCPP) project in Minnesota, Michigan, and Wisconsin to assist NRCS with landowner outreach, coordination, and technical assistance to benefit Golden-winged Warbler, American Woodcock (*Scalopax minor*), and associated species. To date, between the Appalachians and Great Lakes efforts, hundreds of private landowners have work with NRCS and its partners to create Golden-winged Warbler nesting habitat on their properties in (MN, WI, PA, NJ, WV, and MD). Additionally, public land managers in several of these states have also created much needed early successional habitat. In 2015, Indiana University of Pennsylvania and Cornell University began a collaborative effort to monitor avian and vegetation characteristics on private lands enrolled in NRCS conservation programs and nearby public lands (Part 1). In 2016, our partnership initiated a pilot study to fine tune pollinator survey techniques (McNeil et al. 2018) and in 2018, we implemented these pollinator surveys on several of the Golden-winged warbler habitat sites on private and public lands in Pennsylvania (Part 2). Herein, we describe methods and some preliminary results for each project concerning the habitat management initiatives that target habitat for Golden-winged Warbler. These include 1) monitoring and analysis of Golden-winged Warbler response to habitat management across both regions; 2) monitoring and analysis of pollinator diversity and abundance in Pennsylvania. The monitoring efforts outlined here are ultimately essential to help ensure an effective, and ever-evolving, long-term conservation strategy for creating and maintaining breeding season habitat for the Golden-winged Warbler, and to understand the benefits to associated taxa.

## **Part I. Monitoring and evaluating Golden-winged Warbler response to habitat management associated with NRCS conservation programs *Working Lands for Wildlife* (WLFW) and *Regional Conservation Partnership Program* (RCPP)**

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### ***Introduction***

The primary goal of our biological survey effort is to initiate a long-term inventory and monitoring program for Golden-winged Warblers and associated bird species across properties enrolled in NRCS conservation programs (*e.g.*, WLFW, EQIP-Wildlife, etc.) and on lands managed by partner agencies. This year (2018) was the fourth year of this effort and we focused

survey efforts within Pennsylvania, Maryland, New Jersey, West Virginia, and Minnesota. This effort builds on a previous (2012-2014) project funded by NRCS-CEAP (Project ID#: 68-7482-12-502) that quantified and compared several Golden-winged Warbler demographic parameters (*i.e.*, nest success, territory size) among NRCS conservation practices.

Standardized monitoring protocols are used across all states included in this project such that basic demographic data (*e.g.*, singing male densities) and relevant habitat features (*e.g.*, residual trees, shrub/sapling cover, and herbaceous cover) can be consistently collected and compared across all managed sites on participating public lands or private lands enrolled in NRCS programs. Monitoring within areas where habitat management has occurred using standard protocols will provide NRCS staff, public land managers, and their partners with an empirical evaluation of how focal species are benefiting from public and private land management efforts. Information derived from this project combined with conservation practice-specific Golden-winged Warbler demographic parameters collected during the CEAP-GWWA Phase I will inform future conservation planning and potential modifications to existing conservation practice guidelines that target Golden-winged Warblers. These data are the basis for the first multi-year, broad-scale attempt to quantify avian response to recent NRCS-funded private lands conservation programs and similar efforts on public lands in the eastern U.S.

### ***Objectives***

1. Quantify Golden-winged Warbler occupancy and density in areas enrolled in NRCS programs and on public lands in key focal states (*e.g.*, PA, NJ, MD, WV, WI, and MN)
2. Relate avian survey data to site-level vegetation and to use these findings to inform potential modifications to NRCS ranking criteria or other aspects of program delivery.
3. Relate avian survey data to local landscape conditions and to use these findings to inform potential modifications to NRCS ranking criteria or other aspects of program delivery.

### ***Methods***

#### ***Point placement***

We used the ‘*create random points*’ function in the geographic information system, ArcGIS, to generate point locations for vegetation sampling and associated avian monitoring (point count locations). Whenever possible, we placed survey locations at least 80 m from an unmanaged forest edge. We did this to maximize the amount of each treated area sampled. Due to the irregular shape/size of some managed patches, survey locations were necessarily <80 m from an untreated edge and therefore placed at the center of the patch. These patch centroids were identified using the ‘*calculate geometry*’ feature in ArcGIS.

#### ***Golden-winged Warbler sampling***

To quantify Golden-winged Warbler (and associated songbird) use of sites managed using NRCS conservation practices, we conducted passerine point counts from May through June 2015-18. Surveys varied by study region with Appalachian surveys occurring 10 days earlier (15 May-15 June) than within the Great Lakes (25 May – 25 June). These periods were chosen as they correspond to the periods in which most songbird species are at maximum daily

detection probability, including the Golden-winged Warbler. Point counts locations were surveyed twice, annually, for songbirds. Points were each conducted by a single observer during fair weather and took place from 0.5 hr pre-sunrise and continued for 4.5 hours daily. Each point count survey consisted of a 10-minute passive period, followed by a 2-minute Golden-winged Warbler playback, and a final 1-minute passive period. This method is believed to maximize the detection probability for Golden-winged Warblers to nearly 1.0. Still, these data were collected in an occupancy framework to allow for model-based accounts of detection error.

### *Vegetation sampling*

To quantify the microhabitat variables among sites managed using NRCS conservation practices, we conducted a vegetation survey at each point location. We surveyed vegetation from 15 June – 15 July, 2015-18. All vegetation data were collected along three radial transects, each 100 m in length and oriented at 0°, 120°, and 240° from the point count location. Along each transect plant strata measurements were taken at 10 “stops” (10 m apart; n=30/point count location). Vegetation strata recorded at each stop consisted of the presence/absence of sapling, shrub, *Rubus*, fern, forb, sedge/grass, leaf litter, and bare ground. Trees > 10 cm in diameter-at-breast-height were classified as “canopy” and those ≤ 10 cm were considered saplings. Trees were quantified using a basal area prism at the 0m, 50m, and 100m locations along each transect (n=7 total/point). Shrubs were considered woody plants with multiple primary stems (in contrast to single-stemmed saplings). Ferns were seedless vascular plants with compound fronds (*e.g.*, bracken fern, *Pteridium aquilinum*). Forbs were broad-leafed dicotyledonous plants (*e.g.*, *Viola* spp.). The plant category ‘sedge’ included any monocotyledonous plant, however, was frequently *Carex* spp. Plant strata were recorded with an ocular tube such that only strata that intersected with crosshairs in the ocular tube were considered present. While a single stop could include multiple strata types, each stratum could only be represented once/stop and thus each point count location could have a maximum of n=30 occurrences for each stratum. Plant strata values were analyzed as percentages (*i.e.*, % cover) as some sites had outer portions of transects truncated due to irregularly-shaped management boundaries.

### *Occupancy modeling*

We modeled Golden-winged Warbler detections using a series of single-season occupancy models in the R package *unmarked*. We analyzed annual occupancy rates by modeling each year separately. To analyze habitat effects, we stacked all years of data together and analyzed using static occupancy models, accounting for site age as a covariate. For both analyses, we created occupancy models in two tiers: a detection probability tier and a state occupancy tier. The detection tier allowed detection probability to vary as a function of survey covariates (date, weather, etc.). We fit all possible subsets of 0-4 detection covariate using the *dredge* function (package: *MuMin*). Using the top-ranked detection model, we created single-covariate occupancy models in two candidate sets: one for vegetation/microhabitat data (measured in the field) and another for landscape data. Because the Great Lakes dataset was comprised of two management types (shrub management and timber harvest), we modeled the region as a whole and again, separated by management type. Landscape data were extracted from the 2011 National Land Cover Database (NLCD) within 1 km of each sampling location. Models were ranked in accordance to Akaike’s Information Criterion adjusted for small sample size (AIC<sub>c</sub>). The result was AIC<sub>c</sub>-ranked occupancy models that allowed us to assess the impacts of

habitat on Golden-winged Warbler occupancy. We also report the model beta coefficients (85%CI) as a measure of effect size.

## Results

### *Golden-winged Warblers Abundance*

We intend to analyze our data in the future using a distance-sampling approach to generate Golden-winged Warbler density estimates. Such an analysis will compliment our occupancy-based analysis we present below. In Table 1, we provide an initial quantification of Golden-winged Warbler abundance across the habitat sites we monitored during this study.

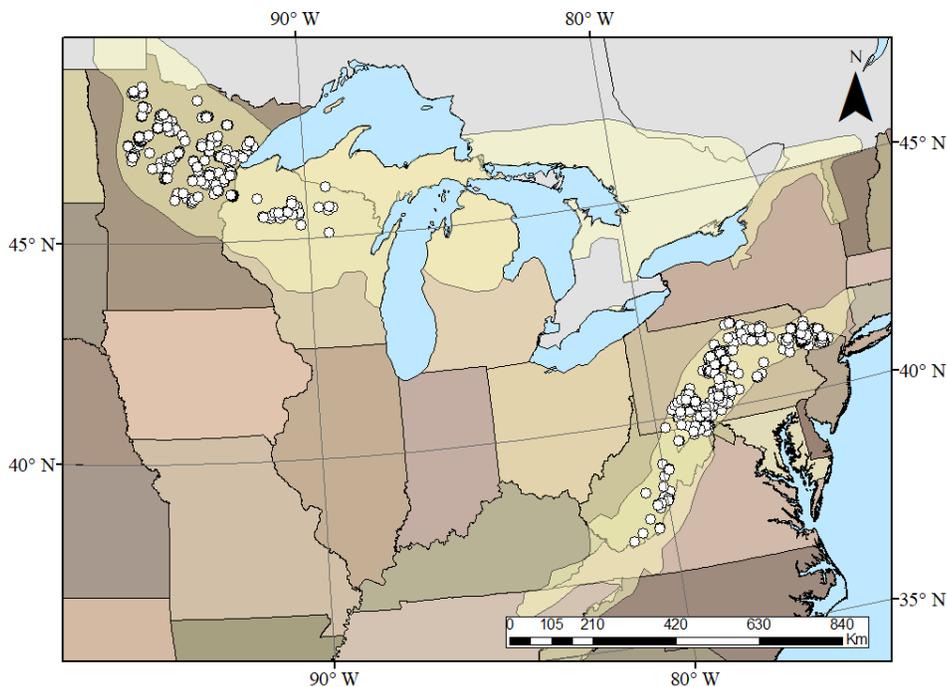
**Table 1.** Golden-winged Warbler counts (abundance) in each habitat x region x year. We report values based on region: Great Lakes (GL) and Appalachians as well as timber harvests and shrublands (GL only). Because Priority Areas for Conservation (PACs) only exist in the Appalachians, only data for those sites are included. N represents the number of point counts sampled.

Treatment	Year	n	Naïve occupancy	Total count	Average # GWWA/point	Average # GWWA/point (within PAC only)
GL-Timber harvest	2015	95	67	120	1.26	—
GL-Shrublands	2015	89	76	143	1.61	—
Appalachian-Timber Harvest	2015	278	56	97	0.35	0.48
GL-Timber harvest	2016	180	135	270	1.50	—
GL-Shrublands	2016	184	164	350	1.90	—
Appalachian-Timber Harvest	2016	425	82	188	0.44	0.64
GL-Timber harvest	2017	213	169	318	1.49	—
GL-Shrublands	2017	202	189	395	1.96	—
Appalachian-Timber Harvest	2017	442	90	131	0.30	0.43
GL-Timber harvest	2018	-	-	-	-	—
GL-Shrublands	2018	177	165	355	2.01	—
Appalachian-Timber Harvest	2018	350	74	110	0.31	0.43

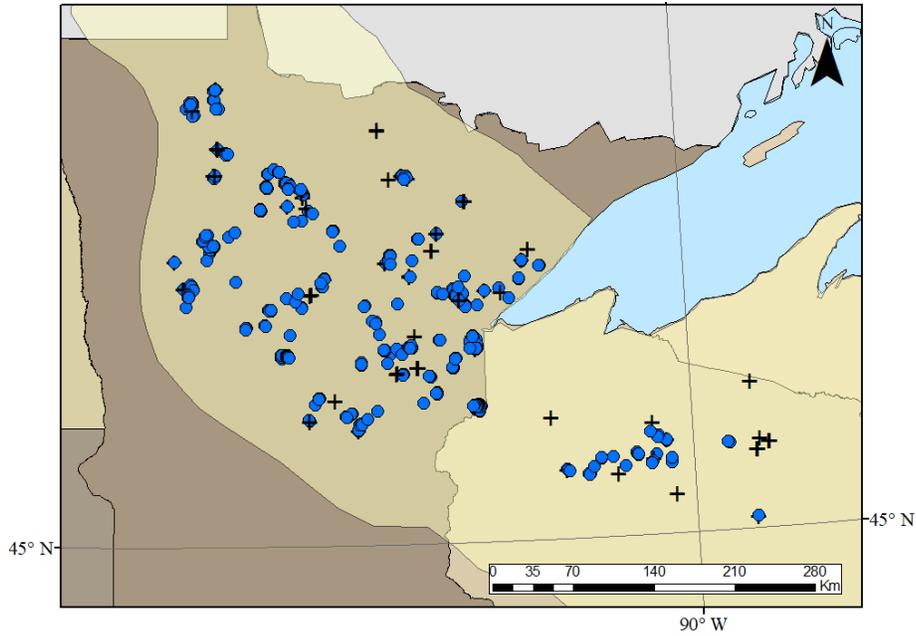
### *Golden-winged Warbler Occupancy*

Across the Appalachian portion of the study area, we conducted avian point counts at 563 locations treated with overstory removal (n=505; 90%) or other BMP-recommended management types (*e.g.*, old field management; n=58; 10%) from 2015-18; Figure 1). These survey locations spanned 31 counties across the states of Pennsylvania, Maryland, New Jersey,

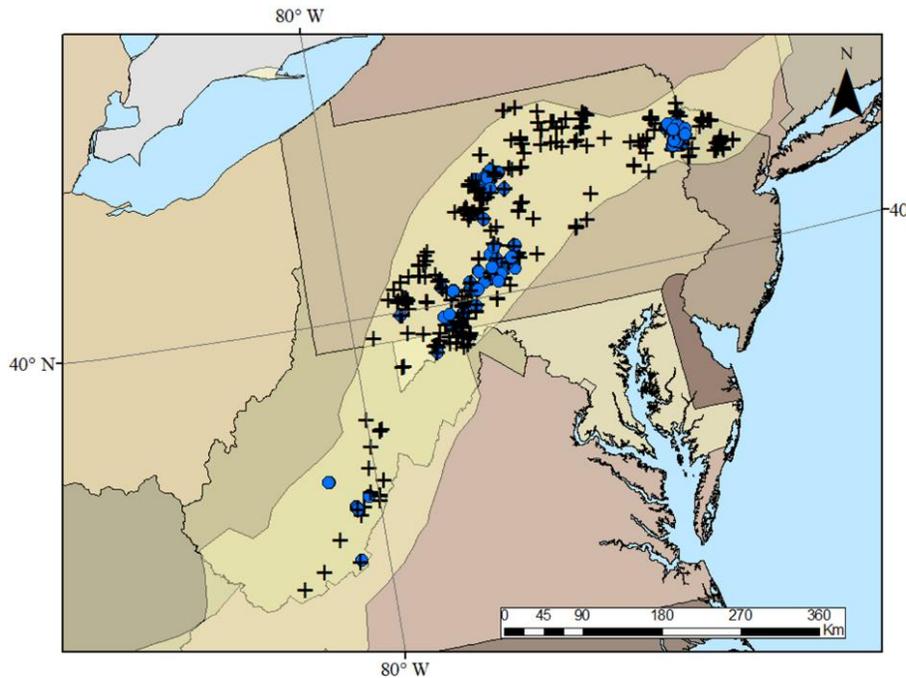
and West Virginia. Among these counties, survey locations were skewed somewhat in favor of private ownership (n=362; 64%) while also sampling many public land sites (n=202; 36%). Appalachian states were not evenly sampled with Pennsylvania receiving more sampling effort than Maryland, West Virginia, or New Jersey (PA: 83%; MD: 6%; WV: 5%; NJ: 6%) due primarily to the limited breeding range of the Golden-winged Warbler within New Jersey and Maryland and limited availability of management sites in West Virginia. Our stacked dataset (all sites, all years) consisted of 1,495 point x year combinations in the Appalachians and 1,141 point x year combinations in the Great Lakes. Over the course of sampling, the number of Golden-winged Warblers observed has steadily increased, particularly on private lands where 14 sites had detections in 2015 to 64 sites with detections in 2018. Detections on private and public lands both appeared to be heavily concentrated within several regions while other portions of the Golden-winged Warbler Appalachian conservation region remained seemingly vacant (*i.e.*, zero detections). Regions with concentrated detections included the Poconos and Pennsylvania Wilds though scattered detections were made across southwestern/south-central Pennsylvania and West Virginia (Fig. 2). Our first Golden-winged Warbler detection in Maryland was made in 2018, but we have yet to observe the species in New Jersey (Fig. 2). Golden-winged Warbler detections were nearly ubiquitous across the Great Lakes region (Fig. 3).



**Figure 1.** Survey locations (shown as white circles) for Golden-winged Warblers and other bird species occurring within early-successional habitats created through NRCS-WLFW, RCPP, and analogous practices on public lands: 2015-18. Surveys were conducted in the Appalachians (Pennsylvania, Maryland, New Jersey, and West Virginia) and the Western Great Lakes (Minnesota, Wisconsin).

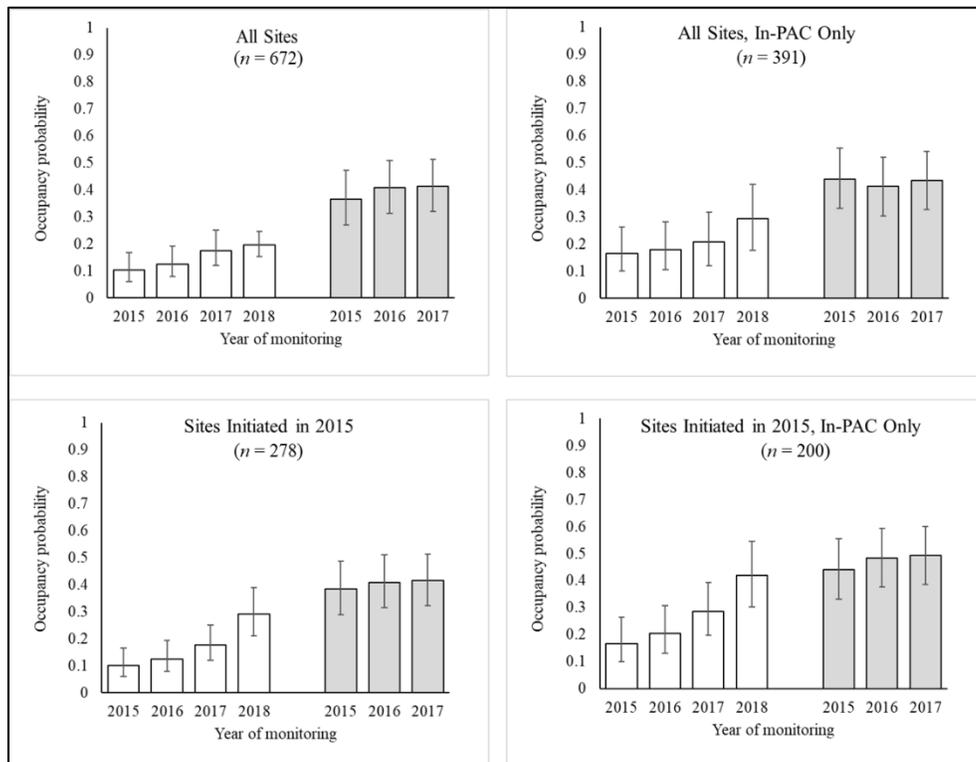


**Figure 2.** Survey locations for Golden-winged Warblers and other bird species occurring within early-successional habitats created through NRCS- RCPP, and analogous practices on public lands: 2015-18 in the Great Lake Region. Locations where Golden-winged Warblers were detected are shown as blue circles while those where the species was never detected are depicted as a “+”. The GWWA Conservation Region is shown as a faded yellow polygon.

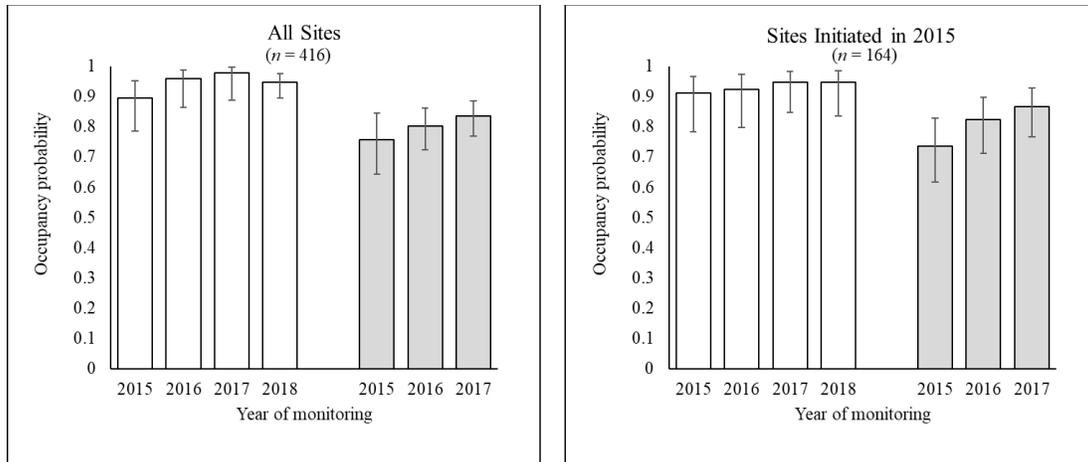


**Figure 3.** Survey locations for Golden-winged Warblers and other bird species occurring within early-successional habitats created through NRCS- WLFW, and analogous practices on public lands: 2015-18 in the Appalachian Mountains Region. Locations where Golden-winged Warblers were detected are shown as blue circles while those where the species was never detected are depicted as a “+”. The GWWA Conservation Region is shown as a faded yellow polygon.

Occupancy models of Golden-winged Warblers within Appalachian sites suggested that detection probability was a function of i) Julian date (negative relationship), ii) time since sunrise (negative relationship) and iii) wind index (negative relationship). Our similar approach in the Great Lakes suggested a negative effect of wind but not Julian date or time since sunrise. That the null model was never well-supported emphasizes the need for a model-based analysis approach such as that which we have applied here. As such, all habitat models (2015-18) for both regions included the relevant terms ('Julian date, time since sunrise, and wind index' and 'wind index'; Appalachian/Great Lakes, respectively) to account for the impact these conditions impose on Golden-winged Warbler detection probability. For sites monitored across all sampling years (private: 2015-18, public: 2015-17), both private and public lands demonstrated consistent increase: private: from 0.10 (95%CI: 0.06-0.17) to 0.20 (95%CI: 15-25) and public: from 0.37 (95%CI: 0.27-0.47) to 0.41 (0.32-0.51; Fig. 4A). This pattern was even more pronounced when restricted to sites monitored within PACs where private lands occupancy increased >250% over four years (Fig. 4B). As expected, this pattern was less pronounced when we do not restrict our analyses to sites monitored across all years because our sample increased substantially during 2016-17 (Fig. 4C-D). A similar pattern was observed within Great Lakes timber harvests, however, this trend was less pronounced for shrub management sites where occupancy rates approached nearly 1.0 by year 2018 (Fig. 5).



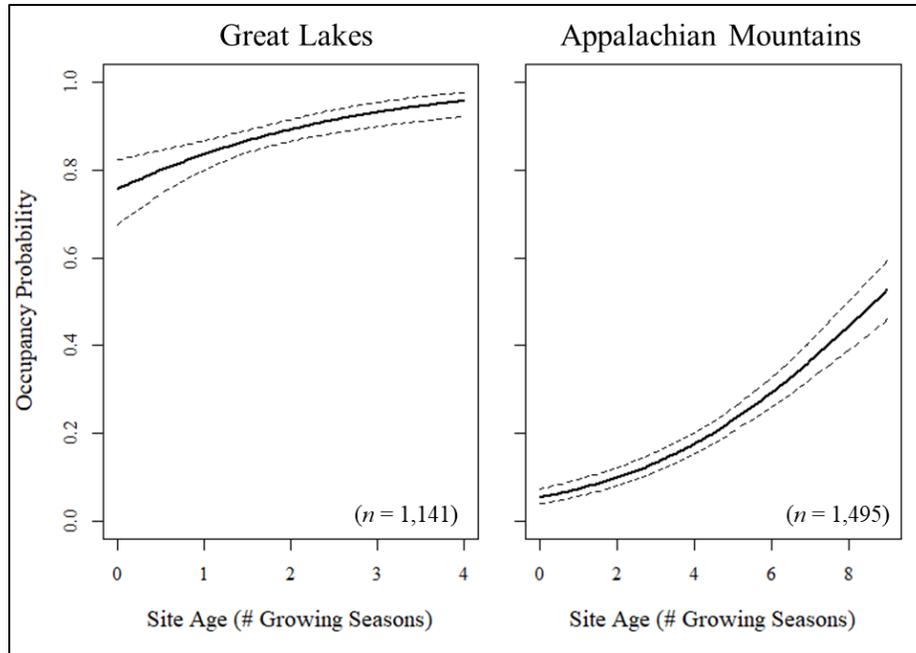
**Figure 4.** Model-estimated occupancy of managed timber harvests monitored across 2015-18 in the Appalachian Mountains. Shown are all sites (top, left), all sites, restricted to PACs (top, right), all sites initiated in 2015 (bottom left), all sites initiated in 2015, restricted to PACs (bottom right). Error bars represent 95% confidence intervals. Sample sizes are reported beneath figure subtitles (*n*). Private timber harvest sites are shown as white bars while public timber harvest sites are shown as gray bars.



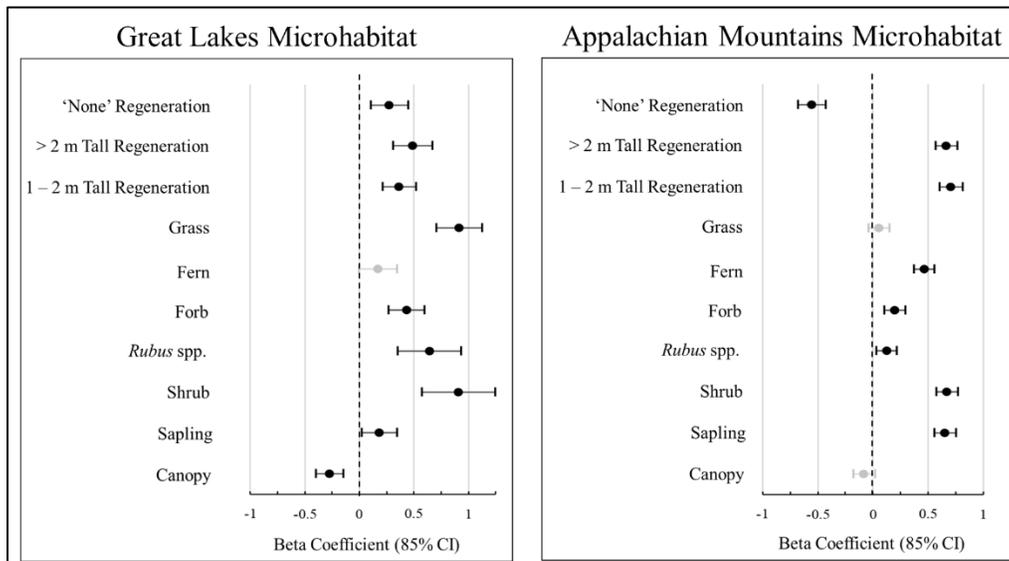
**Figure 5.** Model-estimated occupancy of managed shrublands and timber harvests monitored across the Great Lakes. Shown are all sites (left;  $n = 416$ ) and all sites initiated in 2015 (right;  $n = 164$ ). Note that the sample size for sites monitored in our ‘All Sites’ subset was  $n = 177$  in year 2018. Error bars represent 95% confidence intervals. Shrub management sites (public) are shown as white bars while timber harvest sites (private) are shown as gray bars.

Using data from all sampling years (2015-18) static occupancy models suggested that both the Appalachian- and Great Lakes regions had increasing occupancy as sites aged (Fig. 6). Microhabitat models in the Appalachians suggested that several microhabitat features (all except canopy and grass cover) predicted Golden-winged Warbler occupancy among timber harvests (Fig. 7; Table 3). Similarly, a several microhabitat features (all except fern cover) predicted Golden-winged Warbler occupancy among managed sites in the Great Lakes (Fig. 7). In both regions, sites with more 1-2 m and >2 m woody regeneration had a higher probability of occupancy. Likewise, sites with more forbs, *Rubus*, shrubs, and saplings all hosted higher occupancy probability in both regions (Fig. 7). When we analyzed Great Lakes shrub management- and timber harvest sites separately, we found that most covariates were only weakly associated with occupancy in shrub management sites while the opposite was true in timber harvests (Table 3, Fig. 8).

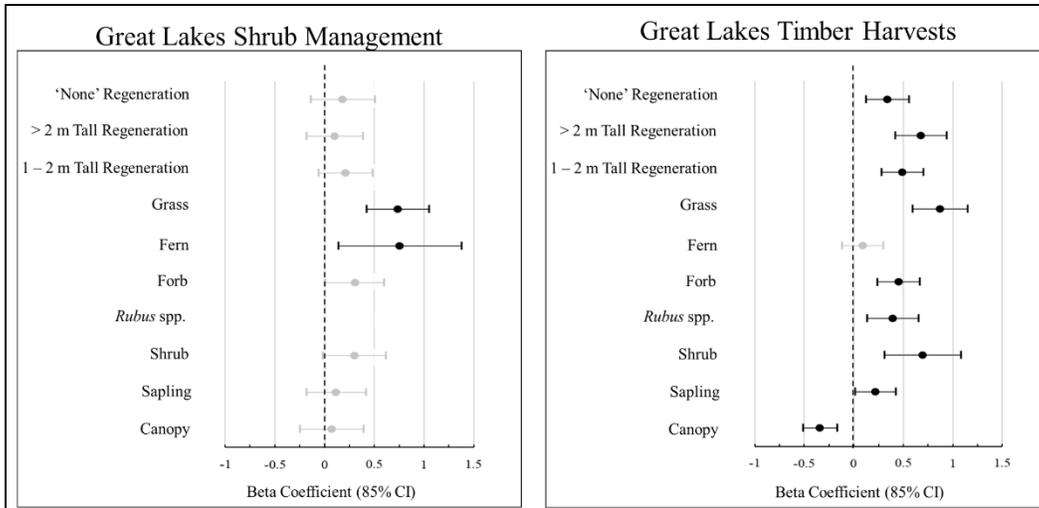
Like microhabitat, we found a variety of landscape-scale features that impacted detection probability in the both regions (Fig. 9; Table 4). In both regions, mixed- and coniferous forests were negatively associated with occupancy while occupancy probability increased as deciduous forest cover increased (Fig. 9). The effects of wetland cover, urban/development, and other landscape covariates largely varied between the two regions. Ongoing analyses will elucidate the effects of forest community type and local metapopulation dynamics on occupancy of these managed sites.



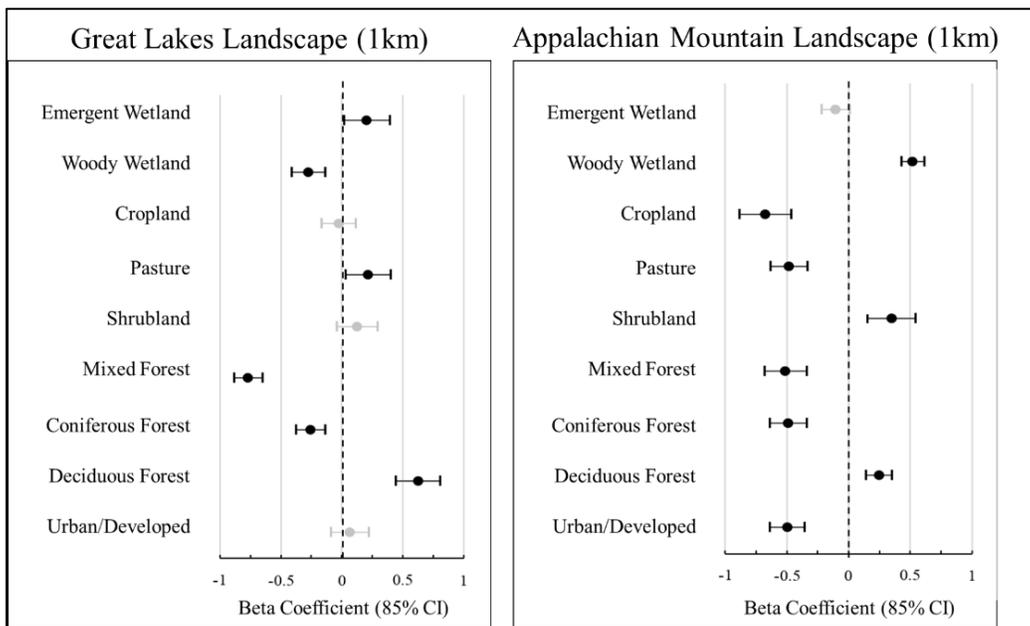
**Figure 6.** Model-estimated occupancy of managed habitats monitored across 2015-18 in the Great Lakes (left) and Appalachian Mountains (right) as a function of site age (number of growing seasons). Dashed lines represent 95% confidence intervals while solid lines represent predicted estimate. Sample sizes are reported beneath figure subtitles ( $n$ )



**Figure 7.** Microhabitat occupancy model beta coefficients for single-covariate static occupancy models in the Great Lakes (left;  $n = 1,141$ ) and Appalachian Mountains (right) from 2015-18. Covariates were derived from 100 m radius vegetation surveys conducted annually at each site. Beta values are depicted as points and 85% confidence intervals are shown as error bars. Those overlapping with zero (dashed vertical line) are interpreted as weak biological relationships (gray: overlapping with zero; black: non-overlapping with 0).



**Figure 8.** Microhabitat occupancy model beta coefficients for single-covariate static occupancy models in Great Lakes shrub management (left;  $n = 652$ ) and timber harvests (right;  $n = 489$ ) from 2015-18. Covariates were derived from 100 m radius vegetation surveys conducted annually at each site. Beta values are depicted as points and 85% confidence intervals are shown as error bars. Those overlapping with zero (dashed vertical line) are interpreted as weak biological relationships (gray: overlapping with zero; black: non-overlapping with 0). *Rubus* spp. was too infrequent in shrub management sites to allow parameter estimation.



**Figure 9.** Landscape-scale (1km radius) occupancy model beta coefficients for single-covariate static occupancy models in the Great Lakes (left) and Appalachian Mountains (right) from 2015-18. Covariates were derived from extraction of National Land Cover Data around sampling points. Beta values are depicted as black points and 85% confidence intervals are shown as error bars. Those overlapping with zero (dashed vertical line) are interpreted as weak biological relationships (gray: overlapping with zero; black: non-overlapping with zero).

**Table 2.** Microhabitat occupancy model rankings for single-covariate static occupancy models in the Great Lakes (top) and Appalachian Mountains (bottom) from 2015-18. Covariates were derived from 100 m radius vegetation surveys conducted annually at each site. Shown are the model (single-covariate), number of parameters (K), Akaike's Information Criterion adjusted for small sample size ( $AIC_c$ ),  $\Delta AIC_c$ , model weight (w), cumulative model weight (Cum.Wt) and log likelihood (LL).

<b>Great Lakes</b>						
<b>Model</b>	<b>K</b>	<b><math>AIC_c</math></b>	<b><math>\Delta AIC_c</math></b>	<b>w</b>	<b>Cum.Wt</b>	<b>LL</b>
$\psi$ (grass cover)	4	2547.50	0.00	1	1	-1269.73
$\psi$ (shrub cover)	4	2572.63	25.14	0	1	-1282.30
$\psi$ (# growing seasons)	4	2584.50	37.01	0	1	-1288.23
$\psi$ ( <i>Rubus</i> cover)	4	2588.86	41.36	0	1	-1290.41
$\psi$ (>2m woody stems [% plots])	4	2589.01	41.52	0	1	-1290.49
$\psi$ (forb cover)	4	2591.74	44.24	0	1	-1291.85
$\psi$ (1-2m woody stems [% plots])	4	2595.37	47.88	0	1	-1293.67
$\psi$ (canopy cover)	4	2598.57	51.07	0	1	-1295.27
$\psi$ ('no' woody stems [% plots])	4	2601.42	53.93	0	1	-1296.69
$\psi$ (sapling cover)	4	2604.53	57.03	0	1	-1298.25
$\psi$ (fern cover)	4	2604.99	57.49	0	1	-1298.48
$\psi$ (.)	3	2605.22	57.72	0	1	-1299.60
<b>Appalachian Mountains</b>						
$\psi$ (# growing seasons)	6	1926.34	0.00	1	1	-957.14
$\psi$ (1-2m woody stems [% plots])	6	1980.56	54.23	0	1	-984.25
$\psi$ (shrub cover)	6	1982.97	56.64	0	1	-985.46
$\psi$ (>2m woody stems [% plots])	6	1984.72	58.38	0	1	-986.33
$\psi$ (sapling cover)	6	1993.17	66.83	0	1	-990.55
$\psi$ (fern cover)	6	2034.30	107.97	0	1	-1011.12
$\psi$ ('no' woody stems [% plots])	6	2037.69	111.35	0	1	-1012.82
$\psi$ (forb cover)	6	2078.23	151.89	0	1	-1033.09
$\psi$ ( <i>Rubus</i> cover)	6	2083.69	157.35	0	1	-1035.82
$\psi$ (.)	5	2085.08	158.74	0	1	-1037.52
$\psi$ (canopy cover)	6	2085.54	159.20	0	1	-1036.74
$\psi$ (grass cover)	6	2086.48	160.14	0	1	-1037.21

**Table 3.** Microhabitat occupancy model rankings for single-covariate static occupancy models in the Great Lakes: shrub management (top) and timber harvest (bottom) from 2015-18. Covariates were derived from 100 m radius vegetation surveys conducted annually at each site. Shown are the model (single-covariate), number of parameters (K), Akaike’s Information Criterion adjusted for small sample size (AIC<sub>c</sub>),  $\Delta$  AIC<sub>c</sub>, model weight (w), cumulative model weight (Cum.Wt) and log likelihood (LL).

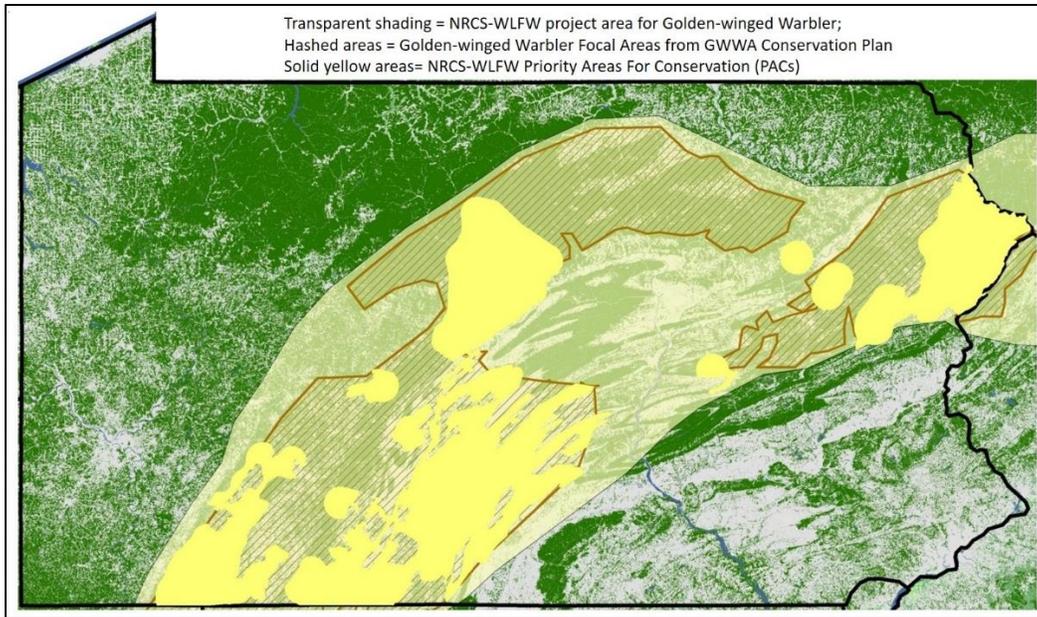
<b>Great Lakes – Shrub Management</b>						
<b>Model</b>	<b>K</b>	<b>AIC<sub>c</sub></b>	<b><math>\Delta</math>AIC<sub>c</sub></b>	<b>w</b>	<b>Cum.Wt</b>	<b>LL</b>
$\psi$ ( <i>Rubus</i> cover)	5	1271.07	0.00	0.61	0.61	-630.49
$\psi$ (grass cover)	5	1272.03	0.96	0.38	0.98	-630.97
$\psi$ (fern cover)	5	1279.19	8.12	0.01	0.99	-634.55
$\psi$ (# growing seasons)	5	1282.68	11.62	0.00	0.99	-636.30
$\psi$ (forb cover)	5	1283.63	12.56	0.00	1.00	-636.77
$\psi$ (shrub cover)	5	1283.83	12.76	0.00	1.00	-636.87
$\psi$ (.)	4	1283.86	12.79	0.00	1.00	-637.90
$\psi$ (1-2m woody stems [% plots])	5	1284.68	13.61	0.00	1.00	-637.29
$\psi$ ('no' woody stems [% plots])	5	1285.17	14.10	0.00	1.00	-637.54
$\psi$ (sapling cover)	5	1285.58	14.51	0.00	1.00	-637.74
$\psi$ (>2m woody stems [% plots])	5	1285.64	14.57	0.00	1.00	-637.77
$\psi$ (canopy cover)	5	1285.79	14.72	0.00	1.00	-637.85
<b>Great Lakes – Timber Harvest</b>						
$\psi$ (grass cover)	5	1185.86	0.00	1.00	1.00	-587.87
$\psi$ (>2m woody stems [% plots])	5	1198.81	12.95	0.00	1.00	-594.34
$\psi$ (shrub cover)	5	1205.28	19.42	0.00	1.00	-597.58
$\psi$ (1-2m woody stems [% plots])	5	1206.05	20.19	0.00	1.00	-597.96
$\psi$ (forb cover)	5	1208.13	22.27	0.00	1.00	-599.00
$\psi$ (# growing seasons)	5	1209.12	23.26	0.00	1.00	-599.50
$\psi$ (canopy cover)	5	1210.52	24.67	0.00	1.00	-600.20
$\psi$ ( <i>Rubus</i> cover)	5	1212.14	26.28	0.00	1.00	-601.01
$\psi$ ('no' woody stems [% plots])	5	1212.57	26.71	0.00	1.00	-601.22
$\psi$ (sapling cover)	5	1215.92	30.06	0.00	1.00	-602.90
$\psi$ (.)	4	1216.30	30.44	0.00	1.00	-604.11
$\psi$ (fern cover)	5	1217.96	32.10	0.00	1.00	-603.92

**Table 4.** Landscape-scale (1 km radius) occupancy model rankings for single-covariate static occupancy models in the Great Lakes (top) and Appalachian Mountains (bottom) from 2015-18. Covariates were derived from extraction of National Land Cover Data around sampling points. Shown are the model (single-covariate), number of parameters (K), Akaike’s Information Criterion adjusted for small sample size (AIC<sub>c</sub>),  $\Delta$  AIC<sub>c</sub>, model weight (w), cumulative model weight (Cum.Wt) and log likelihood (LL).

<b>Great Lakes</b>						
<b>Model</b>	<b>K</b>	<b>AIC<sub>c</sub></b>	<b><math>\Delta</math>AIC<sub>c</sub></b>	<b>w</b>	<b>Cum.Wt</b>	<b>LL</b>
$\psi$ (mixed forest cover)	4	2518.04	0.00	1	1	-1255.00
$\psi$ (deciduous forest cover)	4	2577.68	59.64	0	1	-1284.82
$\psi$ (coniferous forest cover)	4	2598.86	80.82	0	1	-1295.41
$\psi$ (woody wetland cover)	4	2599.33	81.30	0	1	-1295.65
$\psi$ (pasture cover)	4	2603.96	85.93	0	1	-1297.96
$\psi$ (emergent wetland cover)	4	2604.27	86.23	0	1	-1298.12
$\psi$ (.)	3	2605.22	87.18	0	1	-1299.60
$\psi$ (shrubland cover)	4	2605.98	87.95	0	1	-1298.97
$\psi$ (developed land cover)	4	2606.89	88.86	0	1	-1299.43
$\psi$ (cropland cover)	4	2607.16	89.12	0	1	-1299.56
<b>Appalachian Mountains</b>						
$\psi$ (woody wetland cover)	6	2016.74	0.00	1	1	-1002.34
$\psi$ (cropland cover)	6	2048.09	31.35	0	1	-1018.02
$\psi$ (developed land cover)	6	2054.04	37.30	0	1	-1020.99
$\psi$ (coniferous forest cover)	6	2056.15	39.41	0	1	-1022.05
$\psi$ (pasture cover)	6	2057.08	40.33	0	1	-1022.51
$\psi$ (mixed forest cover)	6	2059.18	42.43	0	1	-1023.56
$\psi$ (shrubland cover)	6	2067.46	50.71	0	1	-1027.70
$\psi$ (deciduous forest cover)	6	2075.10	58.35	0	1	-1031.52
$\psi$ (emergent wetland cover)	6	2084.94	68.20	0	1	-1036.44
$\psi$ (.)	5	2085.08	68.33	0	1	-1037.52

### ***Management Implications***

Our results suggest that, to maximize Golden-winged Warbler occupancy of restored habitats in the Appalachian Mountains, managers should focus conservation efforts near pre-existing breeding aggregations (*i.e.*, target sites within PACs; Fig. 10). In contrast, the species was nearly ubiquitous in the Great Lakes suggesting that an analogous strategy may not be needed when management occurs within portions of the Great Lakes Conservation Region we monitored. At landscape scales, patterns also varied by region, but managers should focus efforts within landscapes dominated by deciduous forest with minimal coniferous- and mixed- forest types, regardless of regions. Similarly, stands with the greatest sapling- and shrub cover hosted the highest occupancy rates, regardless of region. In both Conservation Regions, we observed a positive effect of site age (*i.e.*, time since treatment) on occupancy, however, this pattern was most pronounced in the Appalachians where the species is rare. To this end, many Appalachian sites may continue to increase in occupancy while Great Lakes habitats may plateau as occupancy approaches 100%.



**Figure 10.** Priority Areas for Conservation (PACs: solid yellow areas) were delineated using previously collected Golden-winged Warbler occurrence data and other landscape metrics known to influence the species’ distribution (*i.e.*, forest cover). Monitoring efforts on public and private lands have revealed higher occupancy on managed areas within PACs, which are now used by NRCS to rank applications.

**Part II. Habitat Ecology of Native Pollinators within Forested Lands Managed with NRCS conservation programs *Working Lands for Wildlife (WLFW)* and *Regional Conservation Partnership Program (RCPP)***

*Prepared by: Codey Mathis, Indiana University of Pennsylvania; Darin J. McNeil, Jr., Cornell University; Jeffery Larkin, Indiana University of Pennsylvania & American Bird Conservancy*

**Introduction**

Bees and butterflies are functionally important to nearly all terrestrial ecosystems, providing essential pollination services to the majority of extant plant species (Cane 2008; Neff & Simpson 1993). More than 85% of wild flowering plants rely on animal pollination, the majority of which is provided by bees (Ollerton et al. 2011). The pollination services provided by wild insect pollinators are estimated at \$49.1-310.9 million annually (Allsopp et al. 2008). Given the importance of insect pollinators to ecosystem function and service, it is no surprise that their widespread declines have raised alarm (Hallman et al. 2017; Koh et al. 2016; Potts et al. 2010) and many conservation policies have been implemented worldwide to halt and reverse these declines (reviewed by Byrne & Fitzpatrick 2009). Although pollinator population declines are likely driven by a disparate suite of factors (Brown et al. 2016; Goulson et al. 2015), the most important driver is habitat loss (Carman & Jenkins 2016). A recent study found that the Appalachian region of Pennsylvania may have stable populations of wild bees, but there is high

uncertainty with the population estimates (Koh et al. 2016). Moreover, Koh et al. (2016) specified that we need to better understand wild pollinator populations in non-agricultural settings to effectively manage and conserve them.

Eastern North America's forests evolved to be a dynamic mosaic of different forest age classes, where patches were created by natural disturbances and provided floral refuges as they regenerated through ecological succession (Whitney 1994). Today, stands of regenerating forest have become increasingly rare on the landscape as a result of anthropogenic suppression of natural disturbance agents like wildfire and beaver (Askins 2001; DeGraaf & Yamasaki 2003), resulting in a more mature and static forest composition (Askins 2001; Brooks 2003; King & Schlossberg 2014). Still, there is mounting evidence that restoration of early successional forests may provide optimal habitat for stable populations of pollinators (Rivers et al. 2018; Roberts et al. 2017; Winfree et al. 2007).

Habitat management initiatives like *Working Lands for Wildlife* (WLFW) and *Regional Conservation Partnership Program* (RCPP) aim to restore early successional habitat for at-risk wildlife populations (USDA 2014). These initiatives aim to create habitat for imperiled wildlife on private lands across the United States (Cuizio et al. 2013). Within Pennsylvania, there have been more than 4,000 hectares of early successional forest created through WLFW for nesting Golden-Winged Warblers (GWWA; USDA 2014; McNeil et al. 2017). Moreover, the fact that these habitats have already demonstrated to provide habitat for many vertebrate species (McNeil et al. 2018a) further supports the idea that restoration of young forest communities may also provide benefits to insect pollinators. In light of increased implementation of early successional forest management in parts of the Appalachian Mountains, there is an unprecedented opportunity to assess the extent to which bees and butterflies use these habitats. In this study, we examined the ecology of native pollinator populations within early successional forests created through silviculture in the central Appalachian Mountains of Pennsylvania.

### ***Objectives***

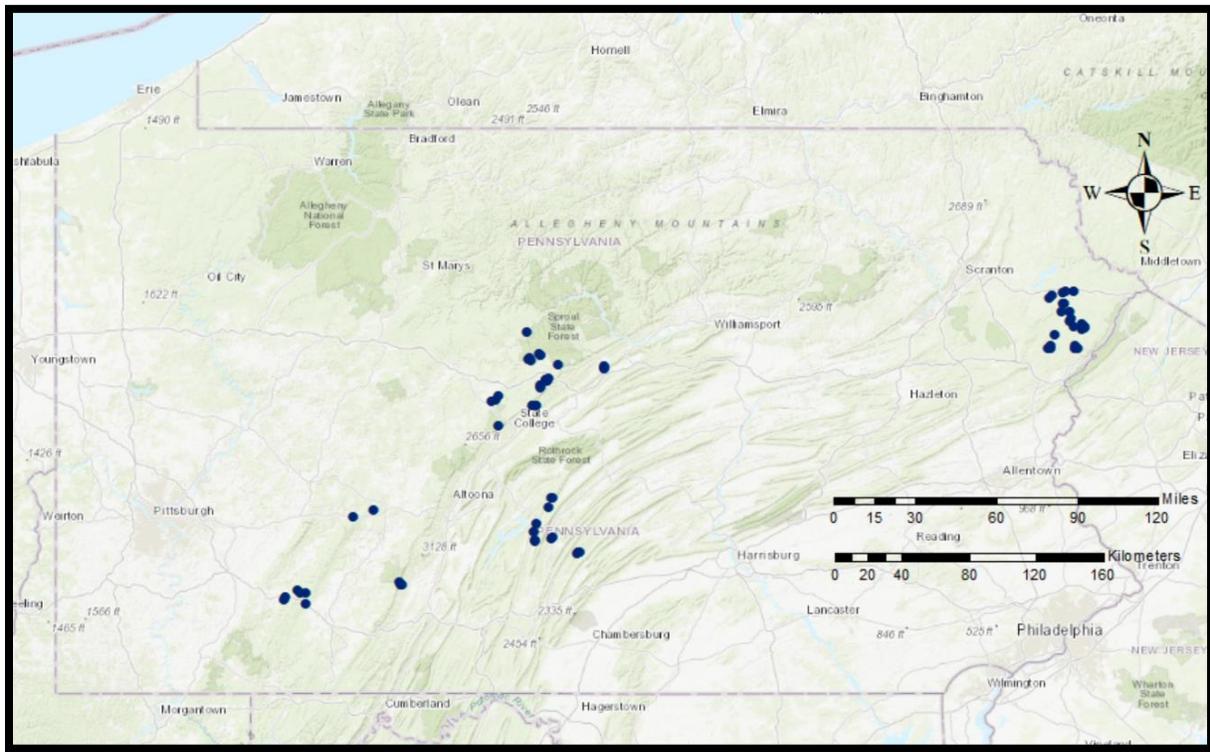
1. Evaluate pollinator use of regeneration timber harvests on public and private lands within portions of the WLFW-GWWA project area of the central Appalachian region.
2. Investigate which structural habitat characteristics drive variation in pollinator abundance within recently harvested early successional forests of the central Appalachian region.
3. Investigate whether stand age and floral resource availability drive variation in pollinator abundance within recently harvested early successional forests of the central Appalachian region.

### ***Methods***

#### ***Study Area***

To investigate native pollinator populations through Pennsylvania, we selected n=75 sites across high-elevation portions of the state, including private (n=38) and public (n=37) lands

managed by regeneration silviculture (Fig. 11). All sites were recently (<10 years) managed through overstory removal (regeneration) harvests. Management of private lands followed conservation plans associated with the USDA-Natural Resource Conservation Services (NRCS) WLFW program. We invited landowners enrolled in WLFW to participate in the pollinator surveys, and our private sites were selected from those who provided access to their properties. Participating private landowners included individual forest tract owners and sportsmanship organizations. We selected public lands that were adjacent to the selected private lands, and those surveyed include State Forests and State Game Lands. Within each site, we place points using an identical protocol as described in Part 1 of this report. We centered a 66 m transect on this point and oriented N-S, which is a modification of the recommendation from the Xerces Society in their bee monitoring protocol (Ward et al. 2014).

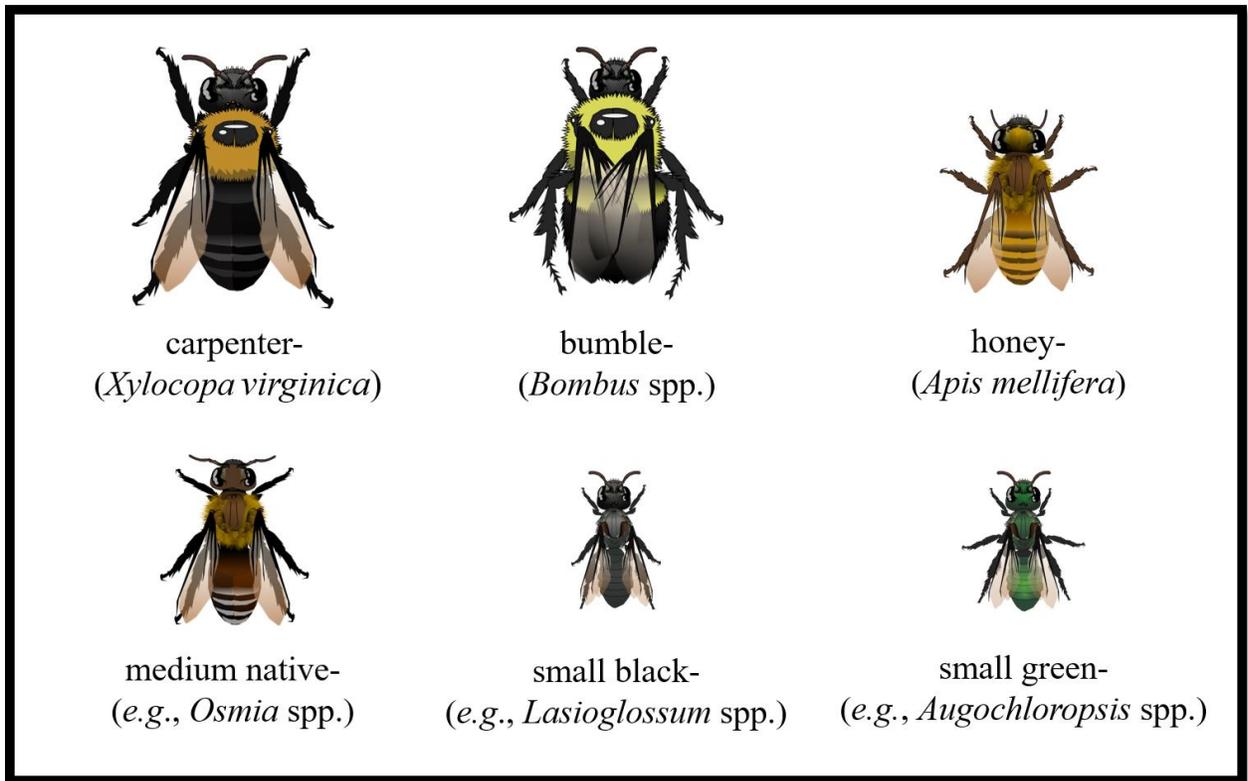


**Figure 11.** A map showing the overstory removal harvests where we surveyed for native bee communities, floral communities, and associated structural habitat characteristics. Note: due to privacy regulations the points shown are not the exact location of study sites.

### *Pollinator Transect Surveys*

We implemented the Xerces Streamlined Bee Monitoring Protocol with distance-sampling protocol adaptations (Ward et al. 2014, McNeil et al. 2018b). During each site visit, a single observer walked the length of the 66m transect for 30 minutes. Given that species- or even genus- level identification of bees often requires a pinned specimen and a microscope (Michener

2007), we identified bees into six groups based on body size and color (Fig. 12): carpenter bees (*Xylocopa virginica*), bumble bees (*Bombus* spp.), honey bees (*Apis mellifera*), medium native bees (e.g. *Osmia* spp.), small black bees (e.g. *Lasioglossum* spp.), and small green bees (e.g. *Agapostemon* spp.). We identified butterflies to species in the field whenever possible, or, when species could not be identified *in situ*, we recorded major identifiable characteristics (e.g., “large, dark swallowtail”). In addition to morphospecies, we also recorded behavior (e.g., resting, flying, feeding) and the estimated perpendicular distance from the transect upon initial detection for each observation. If the pollinator was interacting with a plant, we identified the plant to species. We also recorded covariates for each visit (e.g., wind, cloud cover, temperature). We did not conduct surveys in high winds, rainy conditions, or when the temperature was < 15 degrees Celsius, as these conditions reduce detection probability and fewer pollinators are flying (Ward et al. 2014).

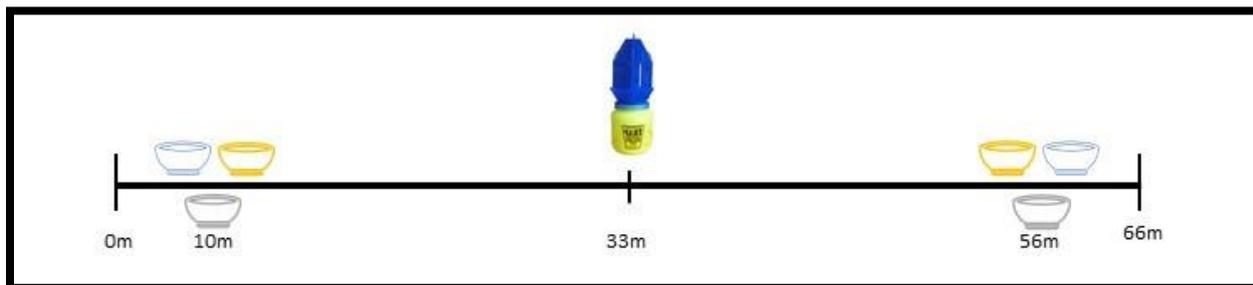


**Figure 12.** Due to the difficulty of identifying bees to species when they are flying, we identified bees to six groups based on body size and color.

### *Pollinator Collection*

In addition to transect surveys, we implemented passive lethal sampling methods to quantify pollinators to the species-level. We passively sampled the pollinator communities on all public lands sites (n = 37) within this study. We placed traps at three locations along each site’s survey transect (Fig. 13). In particular, we used a set of three ground-level bee bowls 23 m from

plot center, North and South. Each set of bowls had a fluorescent yellow, fluorescent blue, and white bowl, which can collect different species of the pollinator community (Droege 2008). In addition to our two sets of bee bowls, we also set a blue-vane trap (SpringStar) at plot center, elevated 1.5 m off the ground using a t-post. We filled all traps with a mixture of Blue Dawn Ultra blue dishwashing soap and water. Trap collection occurred approximately 24 hours after deployment on each site, and we placed collected specimens in vials filled with 70% ethanol and transferred them to a freezer for preservation until they were ready for processing in the laboratory (*i.e.*, pinned, identified, and labeled). We identified each specimen to genus using a stereo microscope and with identification guides for Eastern US bees from Mitchell (1960; vol. 1 and 2).



**Figure 13.** An illustration that shows how the traps were along the 66-m transect. A set of three bee bowls (white, fluorescent yellow, and fluorescent blue) were placed 10m into the transect at either end, and a SpringStar blue-vane trap was elevated 1.5m off the ground at the transect center.

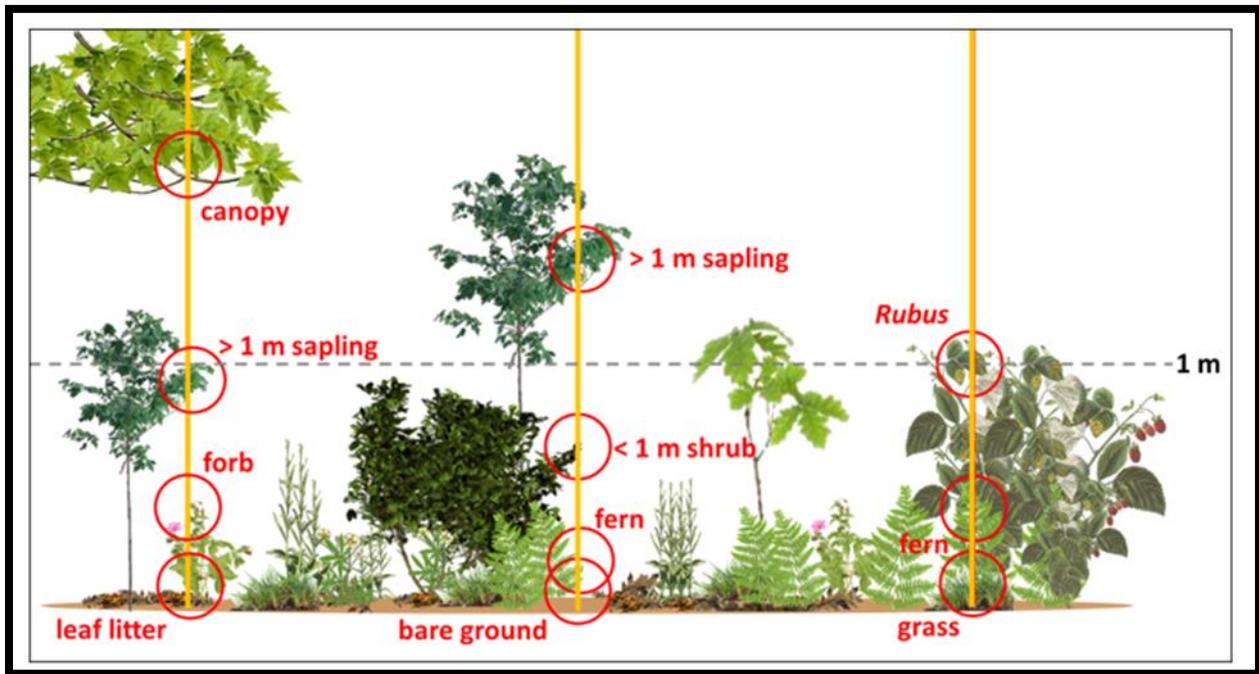
### *Floral Abundance*

Immediately following the pollinator visual surveys, we walked the same transect to quantify: 1) a count of flowering stems and 2) a count of flowers per flowering stem. We defined a ‘flowering stem’ as an individual primary stem (and its associated lateral stems) with any number of flowers upon it, and we identified and recorded each flowering stem within 1m on either side of the transect to species. We differentiated individual stems by connection with the ground – branches occurring *above* the ground were considered part of a single stem while branching *below* the ground created multiple primary stems. We counted individual flowers on each stem when the count was < 20 and estimated counts >20 (to the nearest 10).

### *Site-Level Structural Vegetative Surveys*

We conducted vegetative surveys to quantify vegetation structure within each stand from June-July 2018. Unlike floral resource composition which is expected to vary week-to-week, we sampled vegetation structure only once/site. We collected vegetation data from the survey transect center in 3 radial transects (0 degrees, 120 degrees, and 240 degrees) that were 100 m in

length. We recorded plant strata every 10 m, including presence/absence of saplings, shrubs, *Rubus*, ferns, forbs, sedges, leaf litter, and/or bare ground. We used an ocular tube to record the plant strata, with only the strata observed within the crosshairs of the ocular tube considered present (example: Fig. 14). We defined trees > 10 cm in diameter at breast height (DBH) as ‘canopy’ trees, and those ≤ 10 cm DBH as saplings. A ‘shrub’ was a woody plant with multiple primary stems. A ‘fern’ was a seedless vascular plant with fronds. A ‘forb’ was broad-leafed dicotyledonous plant. A ‘sedge’ was a monocotyledonous plant including plants like sedges, grasses, and rushes. ‘Coarse woody debris’ was any downed woody vegetation like branches and tree trunks. ‘Leaf litter’ was when the view ground was obstructed by a layer of dead leaves, and ‘bare ground’ was when the view of the ground was unobstructed by any vegetation (dead or otherwise).



**Figure 14.** An example of the vegetative survey conducted. Pictured are three vegetation profiles that would be measured using an ocular tube and placed 10 m apart on the transect line. For each tube reading (yellow vertical line), the strata considered ‘present’ are circled and noted in red.

### Statistical analyses

We analyzed our statistics in program R with the packages *unmarked*, *AICmodavg*, and *dplyr*. Using methods defined by McNeil et al. (2018b), we can estimate the density of each pollinator for each site while accounting for imperfect detection of pollinators. To examine which structural habitat characteristics were associated with pollinator abundance, we ran hierarchical distance models on all bees and all butterflies. Each model contained a single habitat

covariate of one of the following: large (>1m) sapling cover, small (<1m) sapling cover, large (>1m) shrub cover, small (<1m) shrub cover, *Rubus* cover, forb cover, fern cover, grass cover, bare ground cover, leaf litter cover, and coarse woody debris cover.

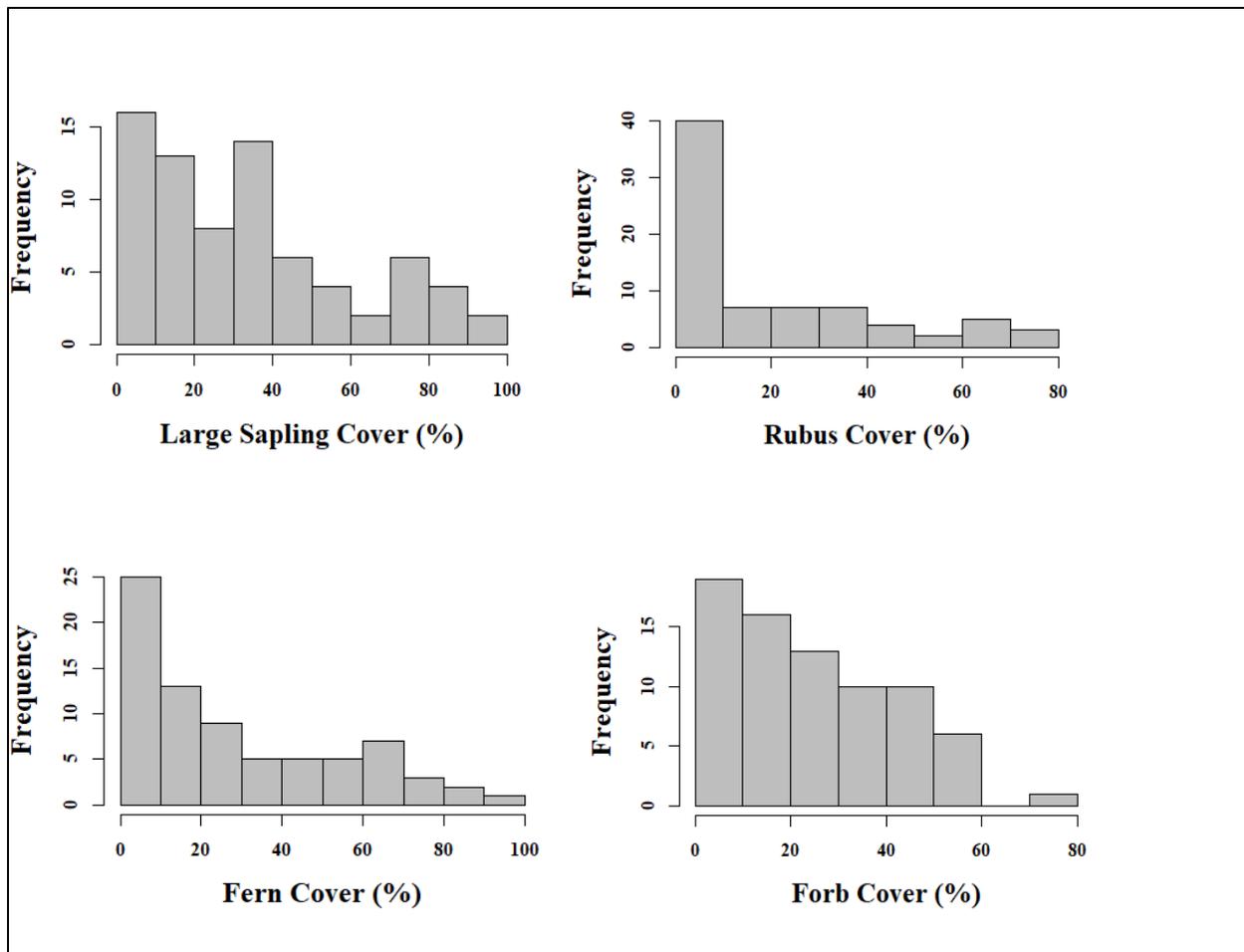
To examine whether stand age and/or floral resource availability drive variation in pollinator abundance, we constructed hierarchical distance models on all bees and all butterflies. We calculated the floral diversity at each site using the Shannon-Wiener Diversity Index ( $H'$ ) (Shannon 1948; Jost 2006). We ran models that accounted for estimated pollinator densities by stand age (years since harvest), floral abundance (count of floral species), and floral diversity ( $H'$ ).

We modeled each visit separately because it would be inappropriate to assume a closed population between each visit. This allowed us to investigate how habitat associations may change for each pollinator over time. For each set of models, we selected models that were more informative than a null (intercept-only) model using Akaike's Information Criterion adjusted for small sample size ( $AIC_c$ ; Burnham and Anderson 2003).

## ***Results***

### *Structural Vegetation within Sites*

We conducted vegetation plots at every site, and our sites ranged from 1-year post-harvest to 9-years post-harvest. Overall, most of our sites contained moderate forb cover and large sapling cover (Fig. 15). In contrast, relatively few sites had high percent cover of *Rubus*, and most sites had low percent cover of ferns. Younger sites tended to have higher small sapling cover, forb cover, and grass cover, given that there is higher light availability for the understory herbaceous layer and the sapling have not had adequate time to grow larger than one meter. Older sites had higher large sapling cover, large shrub cover, and fern cover.



**Figure 15.** Histograms of vegetation features measured on the 100m vegetation transects. Values are expressed as the site-wide average of percent cover for each of the 75 survey locations across the Central Appalachian Region of Pennsylvania.

### *Floral Community*

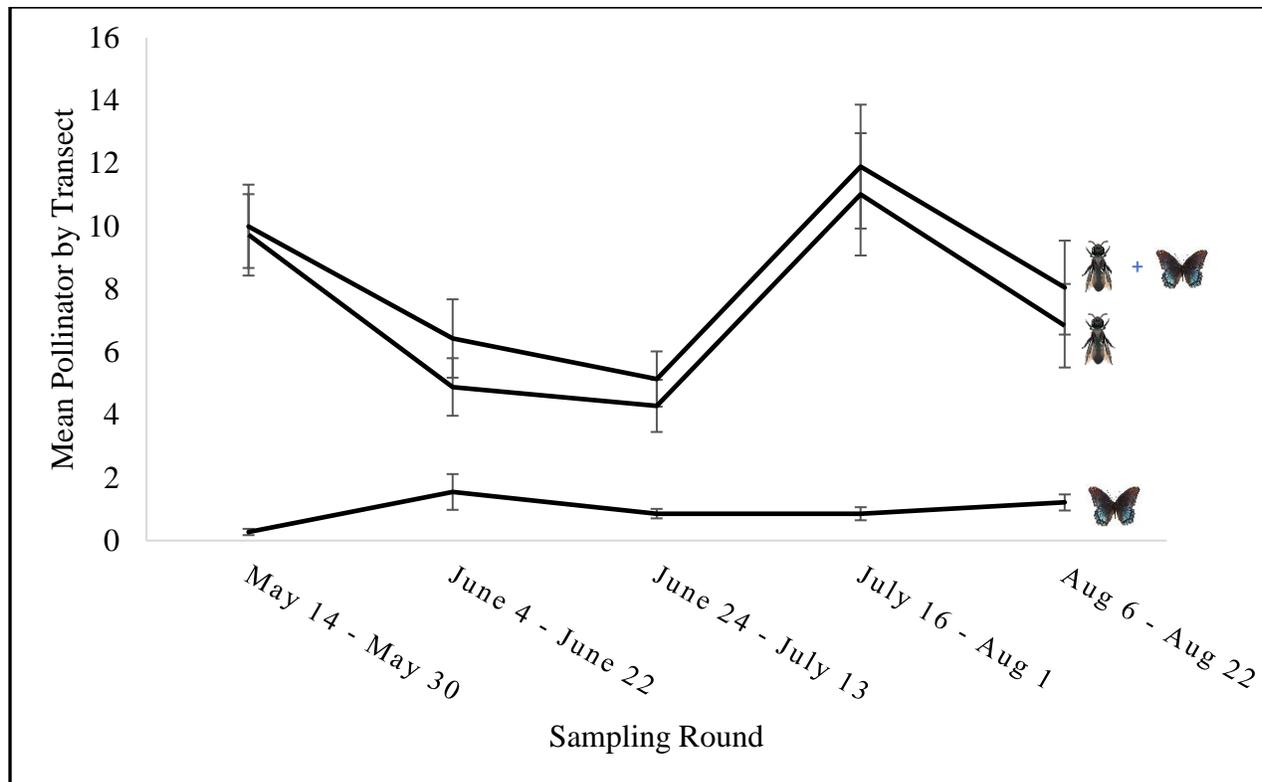
We measured floral resource availability on all sites, estimating a total of >316,000 individual flowers from 165 different taxa over the course of the summer. Most floral resources were available in the first round (May 14-30) because of florally abundant species like black huckleberry (*Gaylussacia baccata*) and various blueberry species (*Vaccinium* sp.). This time of year also falls during peak floral bloom for many flower species, resulting in a higher standardized diversity as well as abundance. As the floral community changed throughout the 2018 season, we observed pollinators using different flower species as they became available. Table 5 includes the top three important floral species for feeding pollinators during each sampling round.

**Table 5.** Flowering plants visited most by pollinators at each of the 75 survey locations across the Central Appalachian Region of Pennsylvania, during 2018.

<b>Round 1 (May 14-May 30)</b>	
lowbush blueberry	<i>Vaccinium angustifolium</i>
black huckleberry	<i>Gaylussacia baccata</i>
highbush blueberry	<i>Vaccinium corymbosum</i>
<b>Round 2 (June 4-June 22)</b>	
Blackberry	<i>Rubus</i> spp.
mountain laurel	<i>Kalmia latifolia</i>
sheep laurel	<i>Kalmia angustifolia</i>
<b>Round 3 (June 24 - July 13)</b>	
Blackberry	<i>Rubus</i> spp.
Northern dewberry	<i>Rubus flagellaris</i>
black cohosh	<i>Actaea racemose</i>
<b>Round 4 (July 16 - Aug 1)</b>	
Blackberry	<i>Rubus</i> spp.
Northern dewberry	<i>Rubus flagellaris</i>
black cohosh	<i>Actaea racemose</i>
<b>Round 5 (Aug 6 - Aug 22)</b>	
white snakeroot	<i>Ageratina altissima</i>
wood-asters	<i>Eurybia</i> sp.
devil's walking stick	<i>Aralia spinosa</i>

### *Pollinator Community*

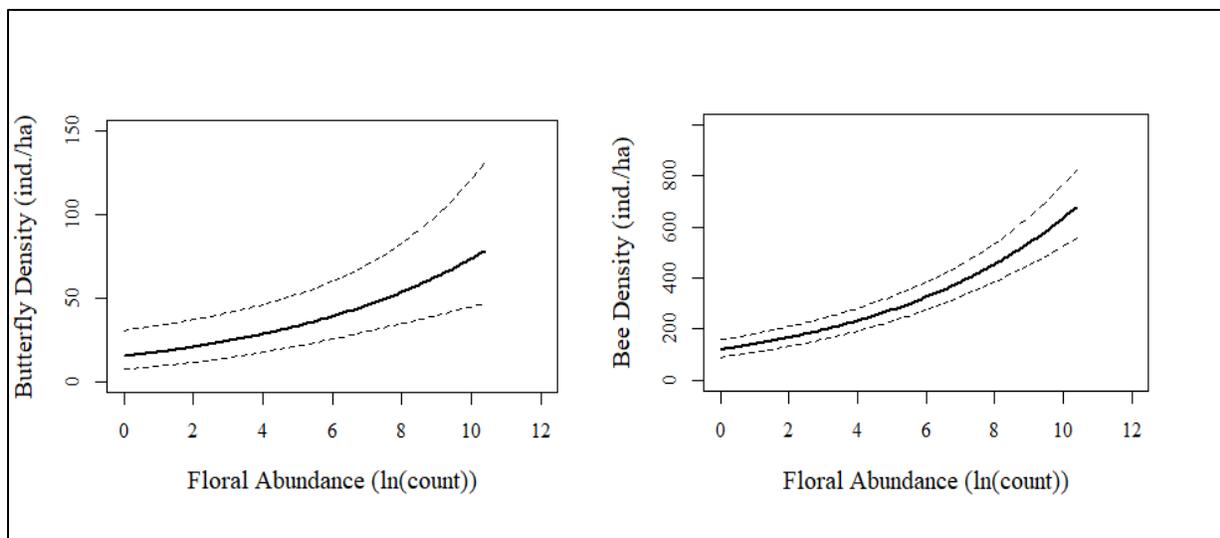
We conducted bee transect surveys at 75 sites every three weeks for a total of five times each (weather permitting). Over these surveys, we detected >2,200 pollinators. Of these, the majority (>1,800) were bees and 279 were butterflies. The most abundant morphospecies that we observed was the small black bee (n=775; 36.6%), followed by the small green bee (n=546; 25.8%) and the butterflies (n=279; 13.2%). We surprisingly observed only 6 carpenter bees over the entire sampling period, and only 21 honeybees. We observed most pollinators in late July (Fig. 16). On public sites (n=37), we collected n=757 bees and n=62 butterflies in our trap arrays across the entire sampling period. The 757 bees we collected represented 22 genera, the most common of which were *Dialictus* (n=218), *Ceratina* (n=209), and *Lasioglossum* (n=62). Identification of specimens to species-level is ongoing.



**Figure 16.** Mean pollinators observed per transect throughout the 2018 field season. We conducted sampling over five rounds, and the different lines correspond to different groups of pollinators: bees + butterflies (combined; top), bees (middle), and butterflies (bottom).

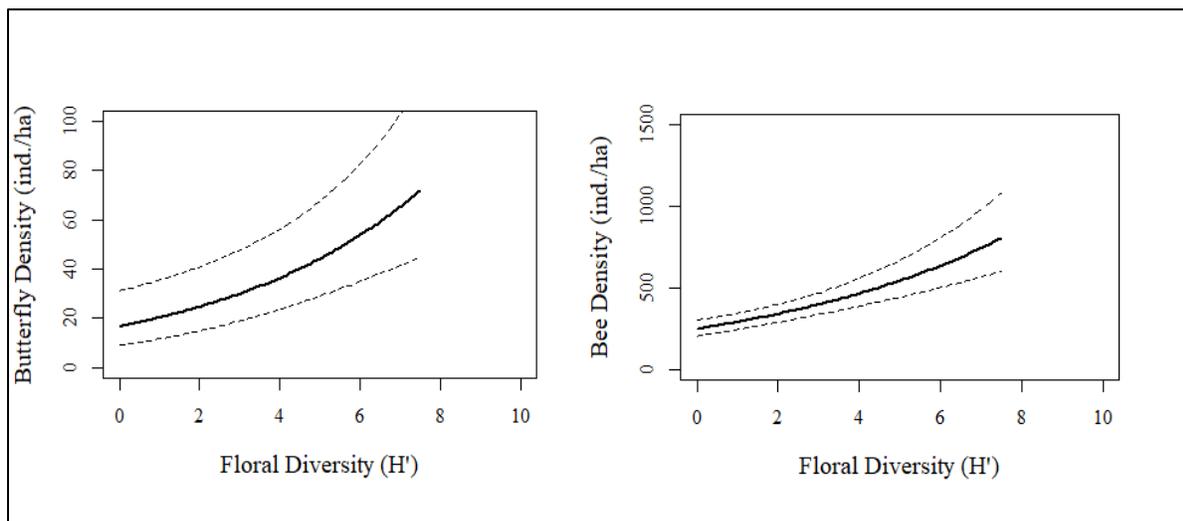
### *Modeling Results*

An abundant floral community is important for species that rely heavily on floral resources to provision their young (Fowler et al. 2016; Roulston & Goodell 2011). We found this to be true for bees across all sampling rounds, but not always true for butterflies (Fig.17; Table 5). This is likely because many butterflies feed on a variety non-floral of foods including fruit and animal scat. Many bees, on the other hand, rely heavily on pollen/nectar as food for both themselves and their developing young (Michener 2008). Our results suggest that, although floral resources are important to both taxa, bees may be more closely tied to floral resource abundance than are butterflies.



**Figure 17.** Estimated pollinator densities (individuals per hectare; butterflies on the left, bees on the right) by the average flowers on a site (log-transformed). The dark line shows our model predictions and the dashed lines depict 95% confidence intervals. The shown models are fit to data from the fifth round of sampling (Aug 2-22, 2018).

Another important component of a healthy floral community is diversity. Our models suggested that average standardized diversity ( $H'$ ) was a significant predictor density for both bees and butterflies (Fig. 18). For example, in the fifth round, our models predict that a site with seven times as many floral species will have pollinator densities four times greater, for both bees and butterflies.



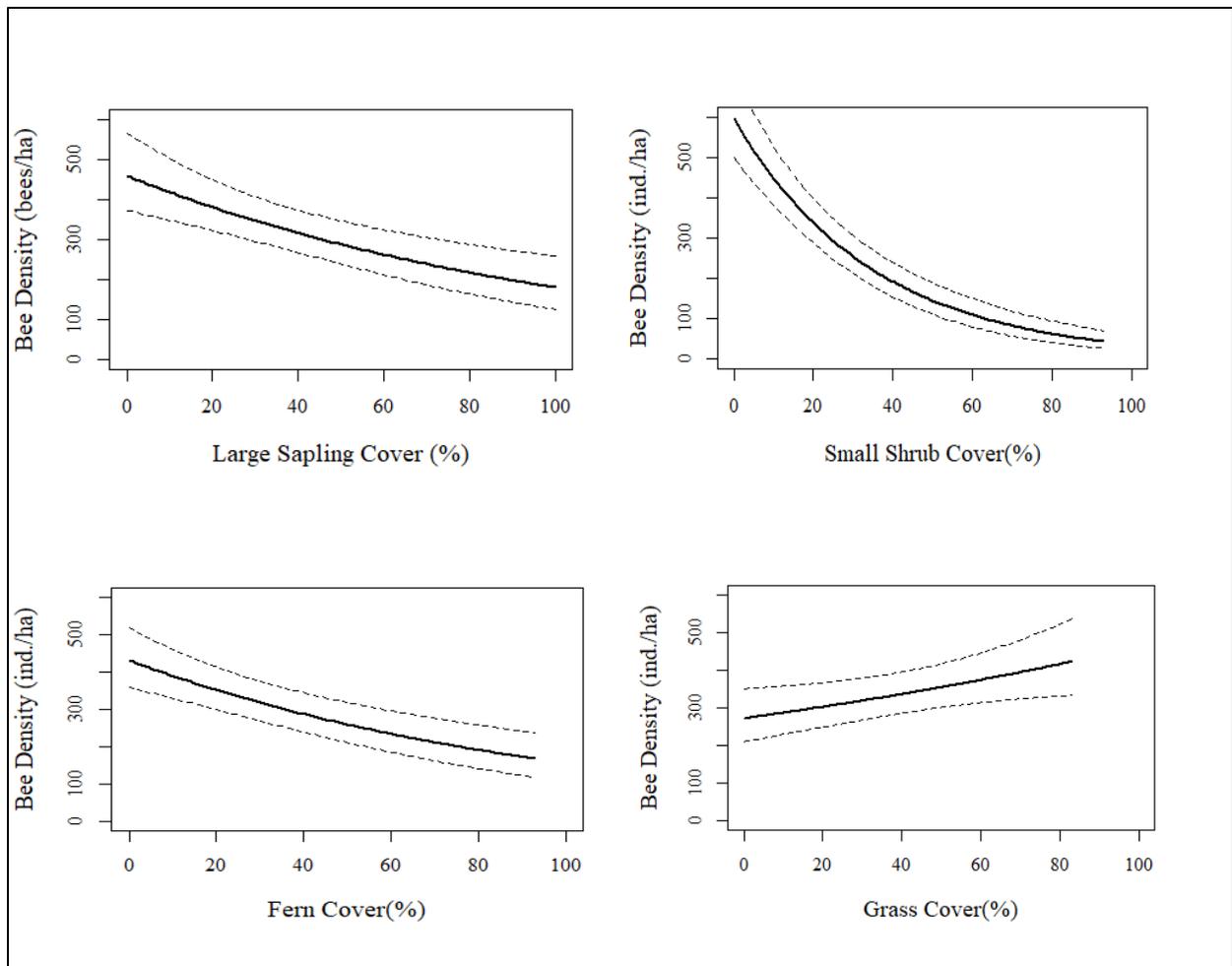
**Figure 18.** Estimated pollinator densities (individuals per hectare; butterflies on the left, bees on the right) by average floral diversity on a site, measured as the standardized Shannon-Wiener Index ( $H'$ ). The dark line shows our model predictions and the dashed lines depict 95% confidence intervals. The shown models are fit to data from the fifth round of sampling (Aug 2-22, 2018).

A main objective of this research is to determine which structural habitat characteristics of regenerating forests landowners can manage for to promote a stable pollinator community on their property. Our models suggest that habitat characteristics associated with pollinator density varies by taxa and, in many cases, are dynamic across a growing season (Table 6).

**Table 6.** Pollinator habitat associations throughout the 2018 Field Season. Survey visit is indicated with ‘V#’ and cell contents (blank, ‘-’, or ‘+’) indicate no-, negative-, or positive association, respectively.

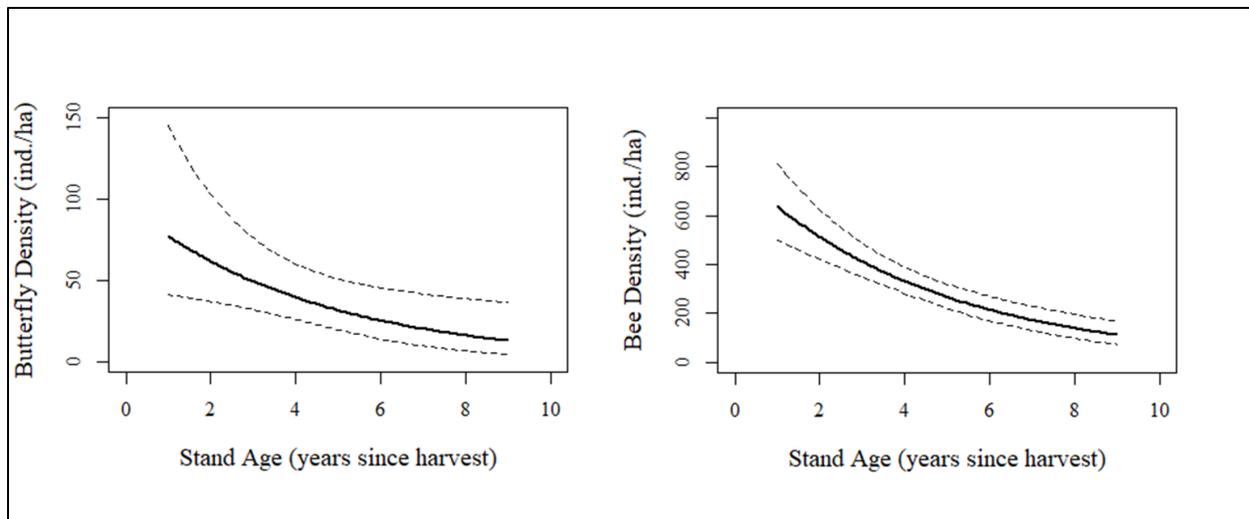
Habitat Variable	Butterflies					Bees				
	V1	V2	V3	V4	V5	V1	V2	V3	V4	V5
Canopy (% cover)			-		-	-	-		-	-
Large (> 1 m) Saplings (% cover)		-				-		-	-	-
Small (< 1 m) Saplings (% cover)		+					+	+	+	+
Large (> 1 m) Shrubs (% cover)		-			-			-	-	-
Small (< 1 m) Shrubs (% cover)				-	-	+	-	-	-	-
Rubus spp. (% cover)					+	-	+	+	+	+
Forbs (% cover)		+				+	+	+	+	+
Ferns (% cover)		-			-	+	-	-	-	-
Grass (% cover)			+	+	+		+	+	+	+
Coarse Woody Debris (% cover)			+				+		+	
Leaf Litter (% cover)		-							+	
Bare Ground (% cover)				-						+
Stand Age (# growing seasons)			-		-		-	-	-	-
Floral Abundance (# flowers)		+	+		+	+	+	+	+	+
Floral Diversity (H')			+		+		+	+	+	+

During most sampling visits, large (>1m) sapling cover and canopy cover were negatively associated with pollinator density (Table 6, Fig. 19). Some notable associations occurred with small (<1m) shrubs and *Rubus* spp. cover. During first visits, most plant-pollinator interactions we observed involved small shrubs like blueberries and black huckleberry (Table 4). As a result, < 1 m shrubs were positively associated with bee density during the visit 1 analysis. However, once those shrubs stopped flowering (visit 2+), density was negatively associated with < 1 m shrub cover, likely because areas with dense shrub cover preclude the growth of other flowering plants via competition. Alternately, *Rubus* spp. was not flowering in the first visit and had a corresponding negative association with bee density. However, once *Rubus* spp. began to flower, we observed a positive association for the rest of the field season. Percent forb cover was an important predictor of pollinator abundance in all sampling rounds. This makes sense because the ‘forb’ category consisted almost entirely of wildflowers that produced floral resources during the course of our sampling (*e.g.*, *Solidago* spp., *Euthamia* spp., *Oxalis* spp., etc.).



**Figure 19.** Vegetation models of bee density as a function of habitat features. All graphs shown include models with slopes discernable from zero. The dark line shows our model predictions and the dashed lines depict 95% confidence intervals. The shown models are fit to data from the fifth round of sampling (Aug 2-22, 2018).

For both bees and butterflies, site age (number of growing seasons, post-harvest) was a strong predictor of density, with oldest sites having the lowest densities (Fig, 20). For example, in the fifth sampling round, our models predict that a 1-year old site hosted five times the abundance of bees as a 9-year old site. Similarly, our models predict that a 1-year site has butterfly communities that are 6 times larger than those on a 9-year site. This makes sense given our aforementioned results, since older stands tend to have taller saplings that reduce light availability for ground-level floral communities, resulting in lower floral abundance and floral diversity within these older stands.



**Figure 20.** Estimated pollinator densities (individuals per hectare; butterflies on the left, bees on the right) as a function of the timber stand age (# of growing seasons). The dark line shows our model predictions and the dashed lines depict 95% confidence intervals. The shown models are fit to data from the fifth round of sampling (Aug 2 – 22, 2018).

### ***Management Implications***

Our results suggest that early-successional habitat created via overstory removal provides habitat for a dense population of pollinators up until 6 years post-harvest. Pollinator density was negatively associated with percent cover of large saplings, shrubs, and canopy, suggesting a mechanism behind the negative effect of age on pollinator densities. As expected, all pollinators benefit from a more diverse floral community. For bees, abundant floral resources seem beneficial, but this pattern is less clear for butterflies. Ensuring an abundance of early-successional habitat (< 6 years post-harvest) within heavily forested landscapes will likely benefit native pollinators within those landscapes. Future activities associated with this project include a) completing identification of specimens collected in 2018 and b) conduct a second year of surveys in 2019 whereby we will survey 100 harvested stands that are less than 6-years old.

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