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Service (NRCS), Regional Conservation Partnership Program (RCPP). The primary focus of this RCPP project was to assist NRCS field offices with delivering outreach and technical and financial assistance to private landowners interested in enhancing forest health to benefit Cerulean Warblers and associated species. Central to the effort was the implementation of conservation practices that align with recommendations found in the management guidelines for enhancing breeding habitat for Cerulean Warblers (Boves et al. 2013; Wood et al 2013). In four years (2015-2019), over 9,606 acres (3,887 ha) of non-industrial, private lands were enrolled in this RCPP. It is important, yet uncommon, that regional scale habitat implementation efforts are accompanied by monitoring programs that allow for rigorous evaluation of biological outcomes (Menz et al., 2013; McNeil et al. *in press*). Such monitoring efforts can provide insight about a program's contribution to species recovery and potential modifications to habitat guidelines or program delivery that may improve biological outcomes (*i.e.*, McNeil et al. *in press*). Herein, we report the preliminary results of avian and vegetation monitoring at sites enrolled in the NRCS's Cerulean Warbler RCPP.

### *Objectives*

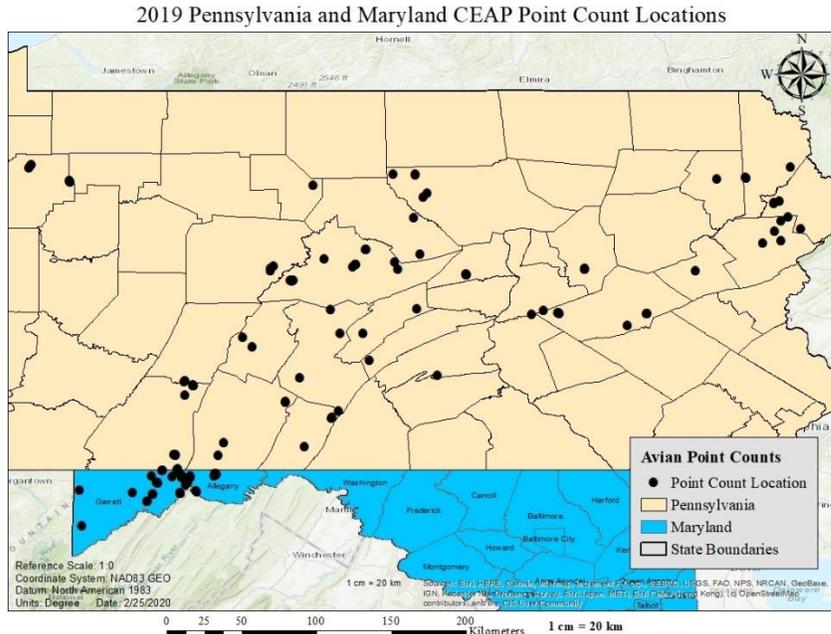
The overall objective of this CEAP component is to evaluate Cerulean Warbler and associated songbird response to shelterwood treatments implemented on private forests in Pennsylvania and western Maryland. Specifically, we aim to

1. Assess Cerulean Warbler response to forest management by quantifying occupancy and density of singing males on private forests enrolled in NRCS programs in Pennsylvania and western Maryland.
2. Relate avian survey data to site-level vegetation and landscape attributes (*i.e.*, aspect, forest type), and to use these findings to inform potential modifications to habitat guidelines, landowner outreach, and conservation delivery.
3. Characterize avian communities associated with forests managed to benefit Cerulean Warbler nesting habitat through NRCS conservation programs in Pennsylvania and Maryland.

## **Methods**

### *Study Area*

We monitored sites in Pennsylvania and western Maryland that were enrolled in NRCS's Cerulean Warbler RCPP (**Fig. 1.1**). All lands enrolled in the project were in heavily forested landscapes (>80% forest cover within 2.4 km (1.5 miles)). All sites were previously treated, or were planned to be, using methods consistent with shelterwood harvests or treatments that reduced canopy cover to some degree (in preparation for additional future harvests). Treated areas ranged in size from 1-21.5 ha and were either unmanaged (pre-management) or 0-4 years post-management. We used the 'create random points' function in the geographic information system, ArcGIS, to generate point locations for avian monitoring (point count locations). Whenever possible, we placed survey locations at least 80 m from the treatment edge. We did this to maximize the amount of each treated area sampled. Due to the irregular size/shape of some habitat 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.



**Figure 1.1.** Locations of avian surveys within managed and unmanaged forests in Pennsylvania and western Maryland during May-June 2017-19. Note: Dots are offset random distances from true locations to protect landowner privacy.

### Avian Monitoring

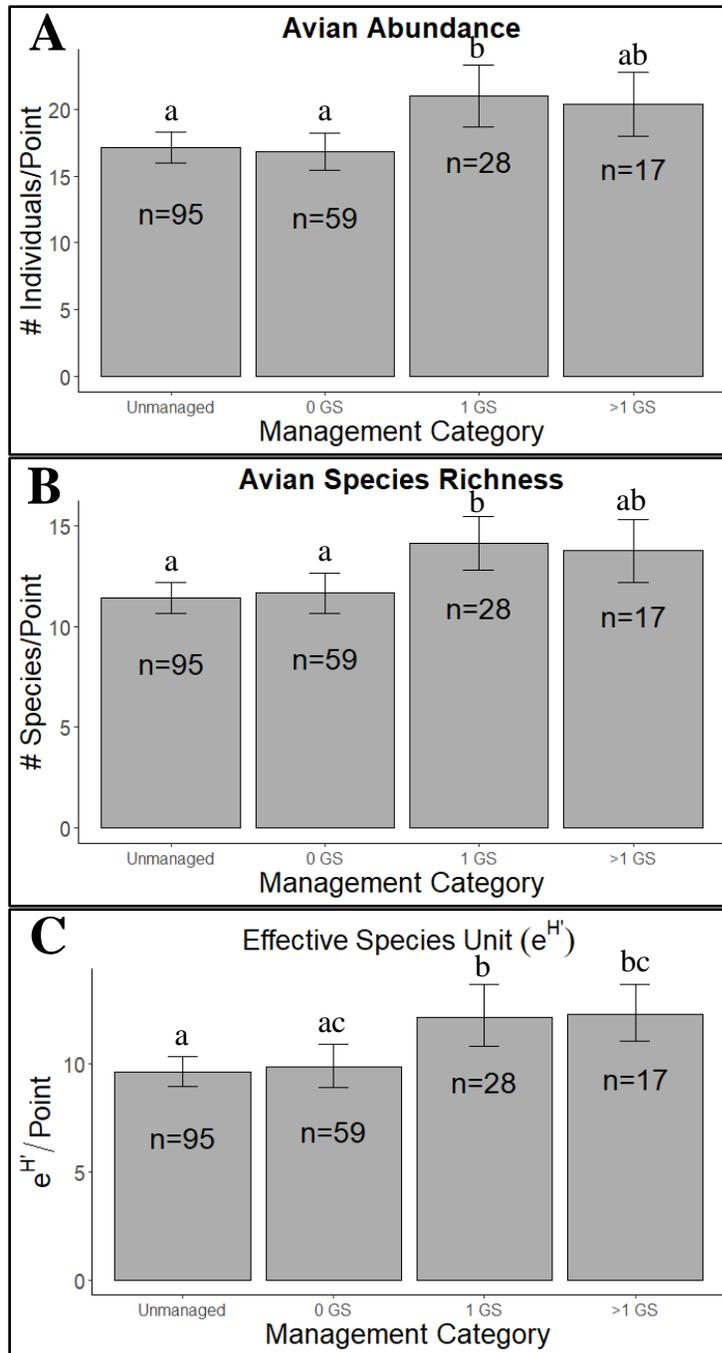
We conducted passerine point counts from 15 May – 15 June 2017-2019 across 28 counties in Pennsylvania and western Maryland. Surveys were conducted between 30-minutes before sunrise and were completed within four hours after sunrise. Each point count survey was 10 minutes in duration and all avian species we detected were recorded. We placed all detections into distance (0-25, 26-50, 51-75, 75-100, and >100m) and time of first detection bins (0-2, 2-4, 4-6, 6-8, and 8-10 mins). We surveyed all point count locations twice annually.

### Vegetation Surveys

We conducted vegetation surveys at all avian point count locations. We used an ocular tube (*i.e.*, GRS densitometer) to estimate cover within the following categories: canopy, fern, forb, grass, leaf litter, bare ground, >1.5 m saplings, <1.5 m saplings, >1.5 m shrub, <1.5 m shrub. We recorded ocular tube readings at every 5m along three transects that extended 35m from point center at 0°, 120° and 240°. We calculated average percent cover for each stratum by summing the total number of presences (1) and dividing that by the total number of times a strata type could have been detected (total possible 21) at each point.

We conducted four wedge prism (10x) readings with one located at point center and the other three located 35m from point center at 0°, 120° and 240° to estimate basal area. We also recorded the species name and measured DBH (diameter at breast height) of all trees and snags considered “in trees” for each prism plot. We categorized crown condition and crown class for all “in trees”. Crown condition was recorded as either full (spreading and healthy), or thin (not full and having gaps). Crown class was recorded as a numeric code indicating the amount of sunlight the crown received based on its position in relationship to the forest canopy. Crown class codes were; “1” (open growth; trees receiving sun from all sides and having no





**Figure 1.2.** Mean avian abundance (A), mean avian species richness (B), and effective species unit (C) for managed and unmanaged forests in western Maryland and Pennsylvania. Surveys ( $n=398$ ) occurred from May-June 2016 across 127 unique point count locations. Bars represent 95% confidence intervals. Different letters denote significant difference based on Tukey post-hoc test.

### Individual Species Abundance

Cerulean Warblers were detected in all four management categories at a total of 14 points (n=18 individuals). Post-management sites (0 GS, 1GS and >1 GS) accounted for more than half (9 of 14; 64%) of Cerulean Warbler occupied points and 72% (13 of 18) of detected individuals. Including Cerulean Warbler, we detected 16 species listed as “species of greatest conservation Need” by the Pennsylvania Wildlife Action Plan (**Table 1.1**).

**Table 1.1.** Species of Greatest Conservation Need detected during surveys in unmanaged and shelterwood harvests in PA and western MD, 2016-19. Included in the table are number of points at which each species was detected, number of individuals detected, and naïve occupancy for each species in each stage (unmanaged, 0 GS, 1 GS, and >1 GS). Bird species common names and the 4-letter banding code can be found at <https://www.pwrc.usgs.gov/bbl/manual/specelist.cfm>

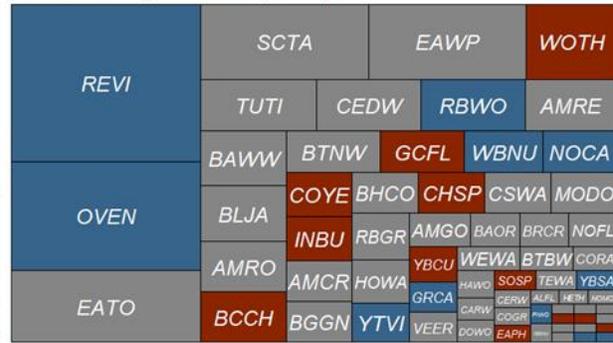
Species	Unmanaged (n=95)			0 GS (n=59)			1 GS (n=28)			>1 GS (n=17)		
	Points	Ind.	Naïve Occ. (%)	Points	Ind.	Naïve Occ. (%)	Points	Ind.	Naïve Occ. (%)	Points	Ind.	Naïve Occ. (%)
<b>BAWW</b>	31	37	32.6	21	23	35.6	11	17	39.3	8	9	47.1*
<b>BLBW</b>	7	8	7.4%	No Detections			3	4	10.7	4	6	23.5*
<b>BRCR</b>	11	11	11.6*	6	8	10.2	2	2	7.7	1	1	5.9
<b>BTBW</b>	10	17	10.5	4	6	6.8	4	6	14.3	3	3	17.6*
<b>BTNW</b>	23	27	24.2*	14	19	23.7	7	9	25.0	4	6	23.5
<b>BWWA</b>	1	2	1.1*	No Detections			No Detections			No Detections		
<b>CERW</b>	5	5	5.3	3	3	5.1	5	8	17.9*	1	2	5.9
<b>EATO</b>	33	54	34.7	37	66	62.7	19	36	67.9	14	24	82.4*
<b>FISP</b>	2	2	2.1	1	1	1.7	1	1	3.6*			
<b>GRCA</b>	16	18	16.8	7	7	11.9	2	2	7.1	5	5	29.4*
<b>GWWA</b>	1	1	1.1*	No Detections			No Detections			No Detections		
<b>HOWA</b>	13	19	13.7	9	12	15.3	7	13	25.0*	4	7	23.5
<b>KEWA</b>	1	1	1.1	1	1	1.7*	No Detections			No Detections		
<b>LOWA</b>	2	2	2.1*	No Detections			No Detections			No Detections		
<b>PRAW</b>	1	1	1.1	No Detections			No Detections			1	1	5.9
<b>SCTA</b>	68	96	71.6	45	61	76.3	20	28	71.4	14	18	82.4*
<b>WOTH</b>	47	85	49.5	21	33	35.6	15	20	53.6*	3	5	17.6

Although four of the five most common species were detected in all four management categories— Red-eyed Vireo (*Vireo olivaceus*), Ovenbird (*Seiurus aurocapilla*), Scarlet Tanager (*Piranga olivacea*), and Eastern Wood Pee-Wee (*Contopus virens*)— gradual shifts in species abundances were observed over time (**Fig. 1.3a-d**). We found significant species trends ( $\geq 10\%$  change in occupancy among the four management categories) for 25 species (**Table 1.2**). Most species (n= 13; 52%) had peak occupancy in management categories with >1 GS and 9 species (36%) had peak occupancy in points with 1 GS. Red-eyed Vireo and Blue Jay occupancy peaked in unmanaged points, whereas Eastern Wood Pee-wee peaked in points with 0 GS. Only 4 of 25 species followed a continuous increasing trend post-management: Eastern Towhee (*Pipilo erythrophthalmus*), Common Yellowthroat (*Geothlypis trichas*), and White-breasted Nuthatch (*Sitta carolinensis*). Red-eyed Vireo was the only species for which occupancy continuously declined post-management.

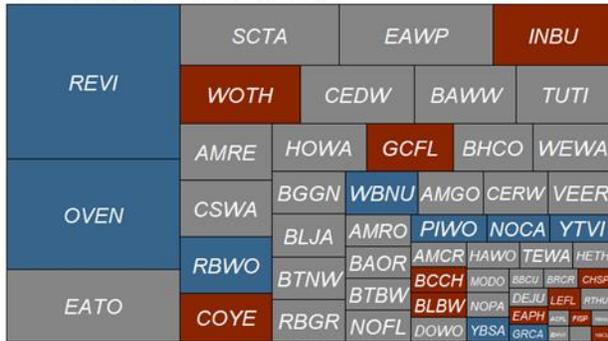
a. Unmanaged (n=95)



b. 0 Growing Season (n=59)



c. 1 Growing Season (n=28)



d. >1 Growing Season (n=17)



**Figure 1.3.** Treemap diagram representing relative species abundance for each management category. Each cell represents a single species, the size of each cell represents the relative abundance of the species, and the color represents the species 10-year population trend in Pennsylvania according to the latest USGS Breeding Bird Survey results Red= decrease; Gray= no change; Blue= increasing. Bird species common names and the 4-letter banding code can be found at <https://www.pwrc.usgs.gov/bbl/manual/speclist.cfm>.





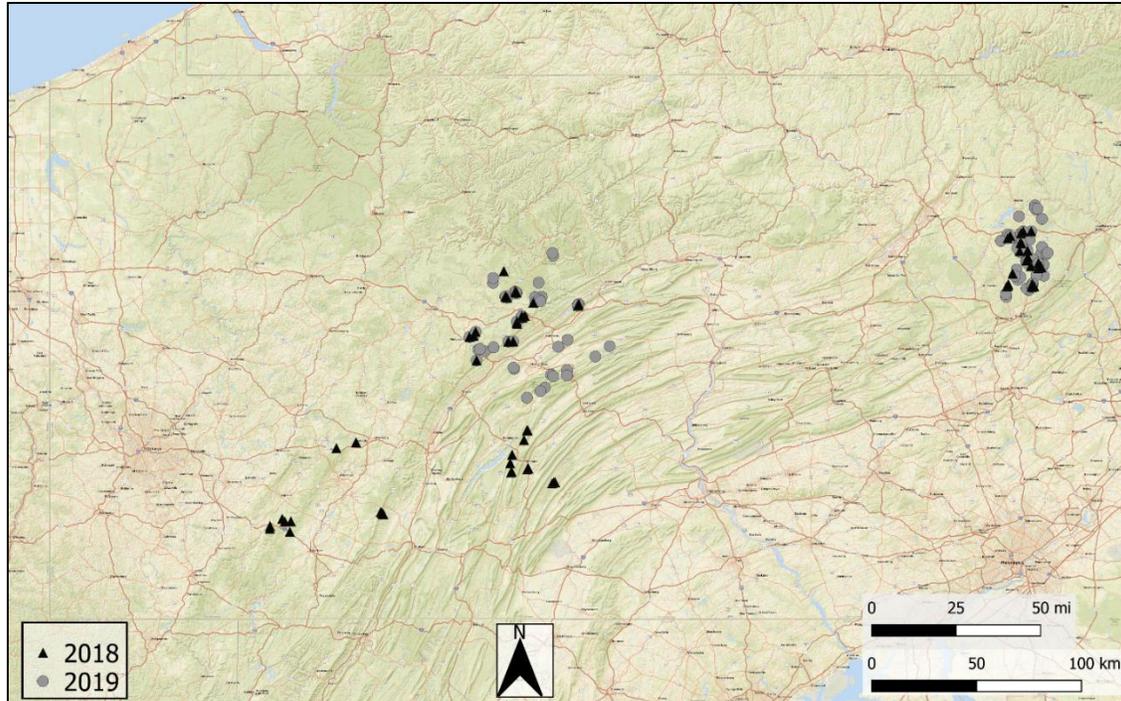








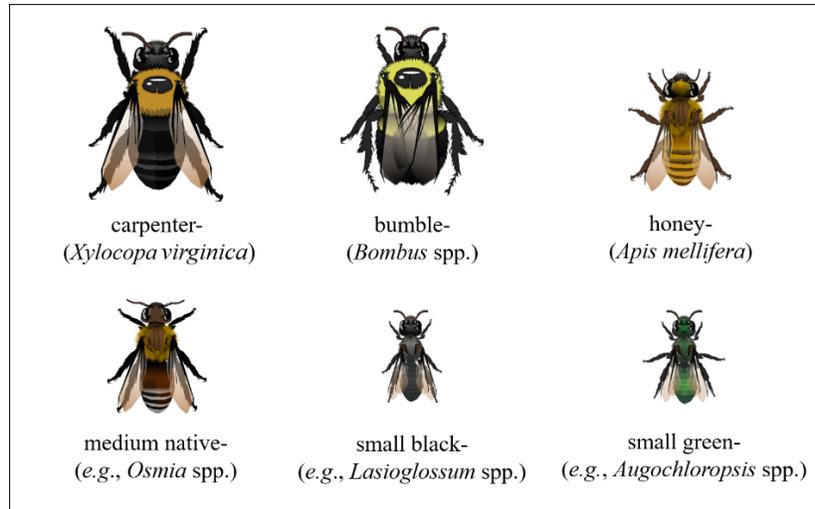




**Figure 2.1.** Map showing the overstory removal harvests where we surveyed for native bee communities, floral communities, and associated structural habitat characteristics. Sites surveyed in 2018 are represented with a black triangle and sites surveyed in 2019 are represented with a grey circle. Some sites were surveyed both years. Note: due to privacy regulations the points shown are not the exact location of study sites.

### *Pollinator Visual Surveys*

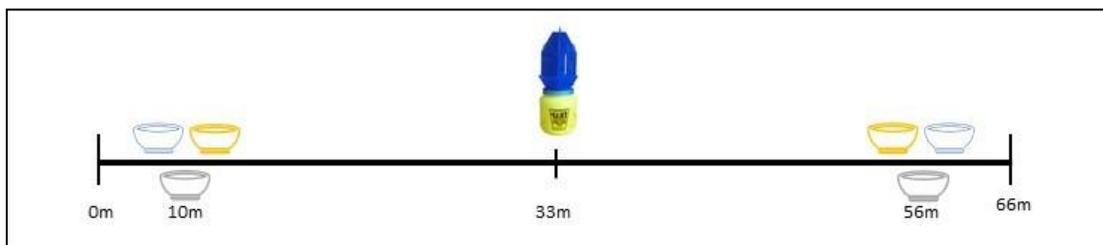
We implemented the Xerces Streamlined Bee Monitoring Protocol with distance-sampling protocol adaptations (Ward et al. 2014; McNeil et al. 2019). During each site visit, a single observer walked the length of the 66m transect for 30 minutes. Given that species- or even genus- level identification of bees often requires a pinned specimen and a microscope (Michener 2007), we identified bees into six ‘morphospecies’ groups based on body size and color (**Fig. 2.2**): 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 survey 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 active (Ward et al. 2014).



**Figure 2.2.** Due to the difficulty of identifying bees to species when they are flying, we identified bees to six ‘morphospecies’ groups based on body size and color.

### Pollinator Specimen 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 on 37 public lands sites in 2018 and 40 randomly selected public (n=20) and private (n=20) lands in 2019. We placed traps at three locations along each site’s survey transect (**Fig. 2.3**). 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 species using a stereo microscope and with identification guides for Eastern US bees from Mitchell (vol.1 1960; vol. 2 1962) and the DiscoverLife identification tools. Species designations were corroborated using specimen collections at the Frost Entomological Museum (Pennsylvania State University, State College, Pennsylvania) and Cornell University Insect Collection (Cornell University, Ithaca, New York).



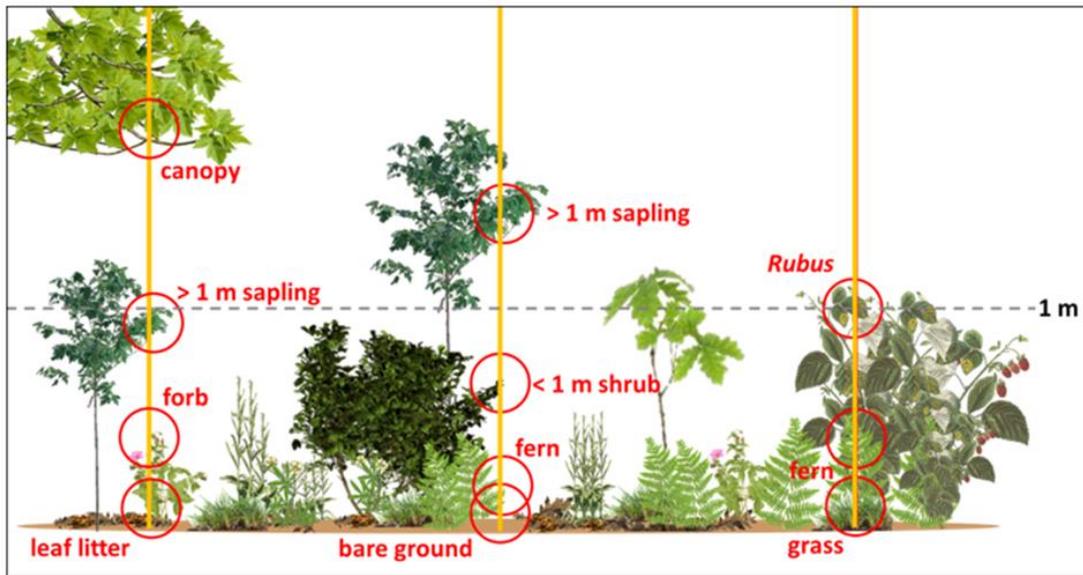
**Figure 2.3.** An illustration that shows trap placement along the 66-m transect. A set of three bee bowls (white, fluorescent yellow, and fluorescent blue) were placed 10m into the transect at either end, and a SpringStar blue-vane trap was elevated 1.5m off the ground at the transect center.

### *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). Identifications were made on-site using assorted floral field guides when possible, or pictures were uploaded to iNaturalist for accurate identification post-survey.

### *Site-Level Structural Vegetative Surveys*

We conducted structural vegetation surveys to quantify vegetation structure within each stand in June/July. Unlike floral resource composition which is expected to vary week-to-week, we sampled vegetation structure only once/site/year. We collected structural 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. 2.4**). We defined trees > 10 cm in diameter at breast height (DBH) as ‘canopy’ trees, and those  $\leq$  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 with a diameter > 10 cm. ‘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 2.4.** 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

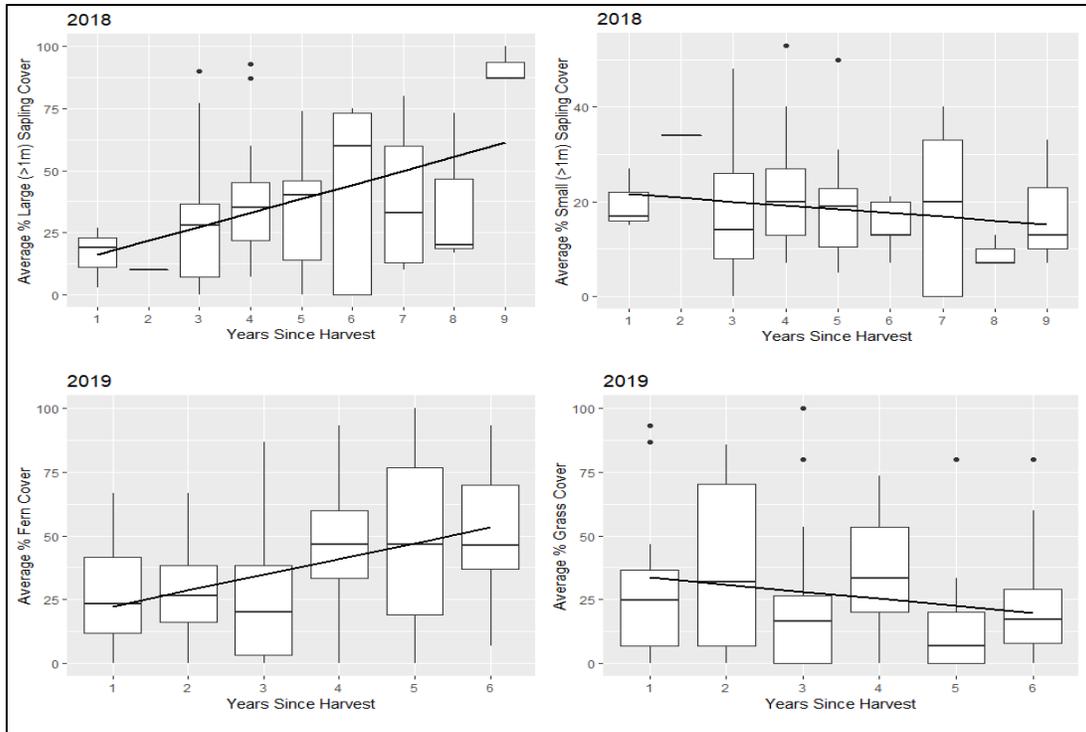
#### Floral Community – “Use” versus Availability

Many studies that have investigated floral communities in relation to pollinator communities have surveyed the floral community in its entirety and did not incorporate any selection of that community for specific floral resources. This occurs because many studies only have collection data and do not include any visual survey methodologies that can infer the behavior of the pollinators associated with the floral communities in the area surrounding the trap. Since we conducted visual surveys that include behavioral associations of pollinators on floral resources, we can determine the floral resources that were available to these communities, as well as the floral resources that they used. We hypothesize that a variable created from floral resources that are selected-for (“used”) would have stronger associations than a variable created from the total floral community (“total”).

First, we determined total floral abundance by summing all flowers for each site and then log-transforming the sums due to non-normality with  $\log(n+1)$ . We then determined the total floral diversity with the Shannon Weiner Index for effective species unit diversity value (Jost 2006). To create the “used” variables, we conducted the same procedure as above with a subset of the total floral resources, but only including the floral species for which we observed pollinators selecting. This subset was determined by looking at the # of surveys in which a floral species was used (through the ‘nectared’ records in the visual survey) / the # of surveys in which a floral species was available. Any species that was used at least 10% of the time that it was available was included in the “used” variable set.



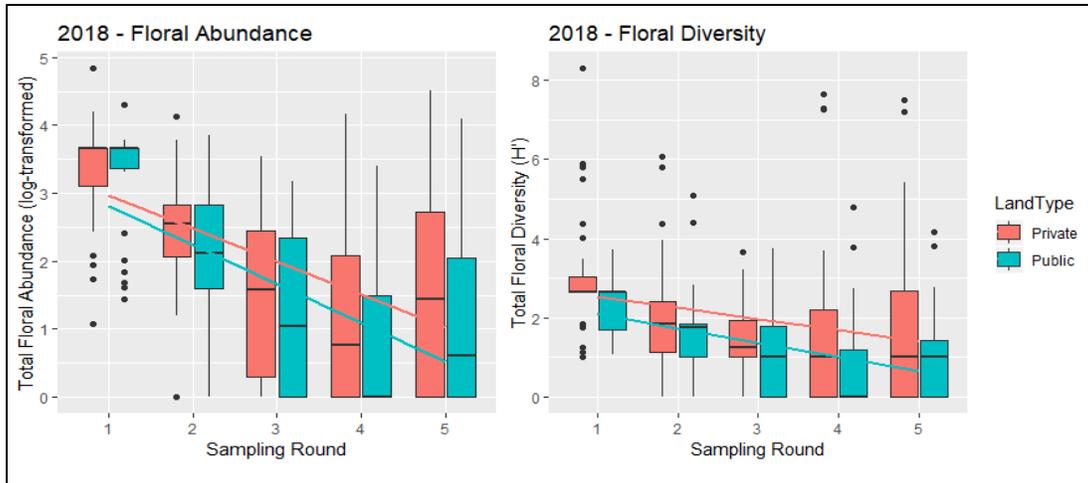




**Figure 2.6.** Boxplots of structural vegetation measured on our sites as a function of stand age (years since harvest), with some 2018 covariates above and some 2019 covariates below. These associations were consistent across years. Values are expressed as the site-wide average of percent cover for each of the survey locations across the Central Appalachian Region of Pennsylvania.

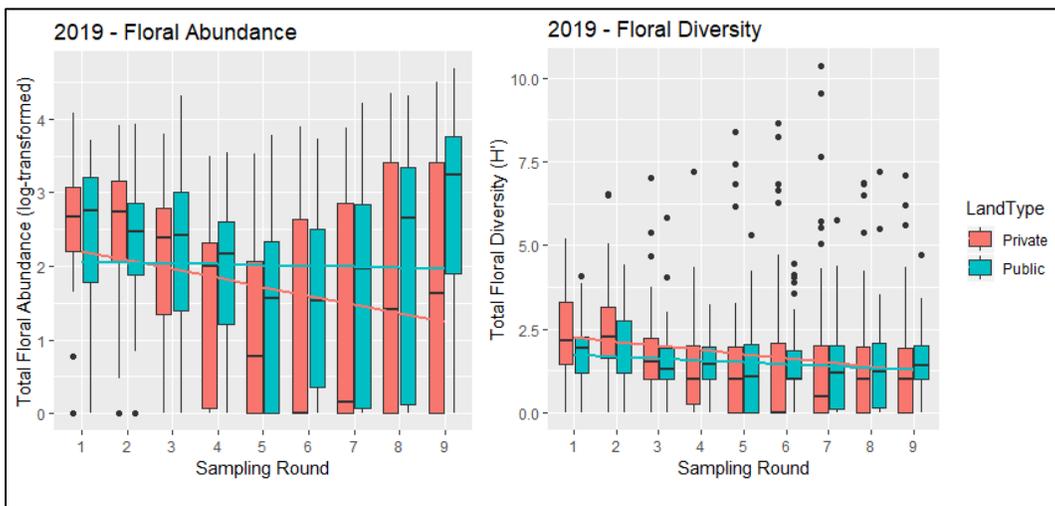
### Floral Community

Both years, we measured floral resource availability on all sites, estimating >1.8 million total individual flowers from over 250 unique species over the course of our surveys. In 2018, 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.; **Fig. 2.7**). This time of year also falls during peak floral bloom for many spring flower species, resulting in a higher standardized diversity as well as abundance. Our 2018 sampling period ended before the fall floral bloom that includes many of the goldenrods (*Solidago* sp.) and fall asters. Privately-owned sites had floral communities that were consistently more abundant and diverse than their public counterparts (**Fig. 2.7**).



**Figure 2.7.** Floral Resources from 2018, grouped by landowner type. Surveys were every 3 weeks from May 14 – August 22, 2018. Floral abundance (left) is expressed as a log-transformation of total floral counts for each site. Floral diversity (right) is expressed as the floral diversity index ( $H'$ ) for each site.

In 2019, we observed a bimodal distribution of floral resources throughout the summer because we fully captured the spring floral bloom (May; mainly blueberries and black huckleberry) as well as the fall floral bloom (August-Sept; goldenrods and asters; **Fig. 2.8**). Privately-owned sites had consistently higher floral diversity until the end of the summer, but public sites had more abundant floral communities (**Fig. 2.8**). We think that this change was because of our focus in 2019 on younger sites ( $\leq 6$  years post-harvest) which tend to have more abundant and diverse floral communities.



**Figure 2.8.** Floral Resources from 2019, grouped by landowner type. Surveys were every 2 weeks from May 14 – September 13, 2019. Floral abundance (left) is expressed as a log-transformation of total floral counts for each site. Floral diversity (right) is expressed as the floral diversity index ( $H'$ ) for each site.

As the floral community changed throughout the summer, we observed pollinators using different flower species as they became available. **Table 2.1** includes the top three important floral species for foraging pollinators during each sampling round.

**Table 2.1.** Flowering plants visited most by pollinators at each of the survey locations across the Central Appalachian Region of Pennsylvania. Rounds are chunked into 3-week intervals based on the timing of the sampling rounds of our 2018 survey efforts.

2018		2019	
<b>Round 1 (May 14 – May 30)</b>		<b>May 14 – May 30</b>	
lowbush blueberry	<i>Vaccinium angustifolium</i>	blueberries	<i>Vaccinium</i> spp.
black huckleberry	<i>Gaylussacia baccata</i>	blackberries	<i>Rubus</i> spp.
highbush blueberry	<i>Vaccinium corymbosum</i>	common cinquefoil	<i>Potentilla simplex</i>
<b>Round 2 (May 31 – June 22)</b>		<b>May 31 – June 22</b>	
Blackberries	<i>Rubus</i> spp.	blackberries	<i>Rubus</i> spp.
mountain laurel	<i>Kalmia latifolia</i>	deerberry	<i>Vaccinium stamineum</i>
sheep laurel	<i>Kalmia angustifolia</i>	multiflora rose	<i>Rosa multiflora</i>
<b>Round 3 (June 23 - July 13)</b>		<b>June 23 – July 13</b>	
blackberries	<i>Rubus</i> spp.	northern dewberry	<i>Rubus flagellaris</i>
northern dewberry	<i>Rubus flagellaris</i>	black huckleberry	<i>Gaylussacia baccata</i>
black cohosh	<i>Actaea racemose</i>	milkweed	<i>Asclepias syriaca</i>
<b>Round 4 (July 14 - Aug 1)</b>		<b>July 14 – Aug 1</b>	
blackberries	<i>Rubus</i> spp.	indian tobacco	<i>Lobelia inflata</i>
northern dewberry	<i>Rubus flagellaris</i>	fireweed	<i>Erechtites hieraciifolius</i>
black cohosh	<i>Actaea racemose</i>	American pokeweed	<i>Phytolacca americana</i>
<b>Round 5 (Aug 2 - Aug 22)</b>		<b>Aug 2 – Aug 22</b>	
white snakeroot	<i>Ageratina altissima</i>	fireweed	<i>Erechtites hieraciifolius</i>
wood-asters	<i>Eurybia</i> sp.	early goldenrod	<i>Solidago juncea</i>
devil's walking stick	<i>Aralia spinosa</i>	flat-top white aster	<i>Doellingeria umbellata</i>
<b>No Survey (Aug 23 – Sept 13)</b>		<b>Aug 23 – Sept 13</b>	
-	-	fireweed	<i>Erechtites hieraciifolius</i>
-	-	wrinkle-leaved goldenrod	<i>Solidago rugosa</i>
-	-	grass-leaved goldenrod	<i>Euthamia graminifolia</i>

### Pollinator Community

In 2018, we conducted visual transect surveys at 75 sites every three weeks for a total of five visits each (weather permitting). Over these surveys, we observed >2,200 pollinators. Of these, the majority (>1,900) 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%). Butterflies made up 13.2% of all observations (n=279). We observed most pollinators in late July (**Fig. 2.9**), and frequently observed more pollinators on privately-owned lands than on nearby publicly owned sites. On public sites (n=37), we collected 775 bees and 57 butterflies in our trap arrays across the entire sampling period. The bees we collected represented 22 genera and 82 species, with the most common genera being *Ceratina* (n=219), *Dialictus* (n=209), and *Augochlorella* (n=55).







*Table 2.3. Butterfly habitat associations throughout the 2019 Field Season. Survey visit is indicated with 'V#' and cell contents (blank, '-', or '+') indicate no-, negative-, or positive association, respectively.*

2019 Butterfly Habitat Associations									
Habitat Variables	V1	V2	V3	V4	V5	V6	V7	V8	V9
Canopy (% cover)		-					-		
Large (> 1 m) Saplings (% cover)				-					-
Small (< 1 m) Saplings (% cover)									
Large (> 1 m) Shrubs (% cover)								-	
Small (< 1 m) Shrubs (% cover)						-			-
<i>Rubus</i> spp. (% cover)		+					+	+	
Forbs (% cover)	+	+		+		+	+	+	+
Ferns (% cover)								-	
Grass (% cover)	+	+	+			+	+	+	+
Coarse Woody Debris (% cover)									
Leaf Litter (% cover)									
Bare Ground (% cover)						+	+	+	+
Stand Age (# growing seasons)							-	-	
Elevation (m)	+							-	
Total Floral Abundance (# flowers)	+	+		+	+	+	+	+	+
Used Floral Abundance (# flowers)	+	+		+	+	+	+	+	+
Total Floral Diversity (H')		+				+	+	+	+
Used Floral Diversity (H')	+	+		+		+	+	+	+

*Table 2.4. Bee habitat associations throughout the 2019 Field Season. Survey visit is indicated with 'V#' and cell contents (blank, '-', or '+') indicate no-, negative-, or positive association, respectively.*

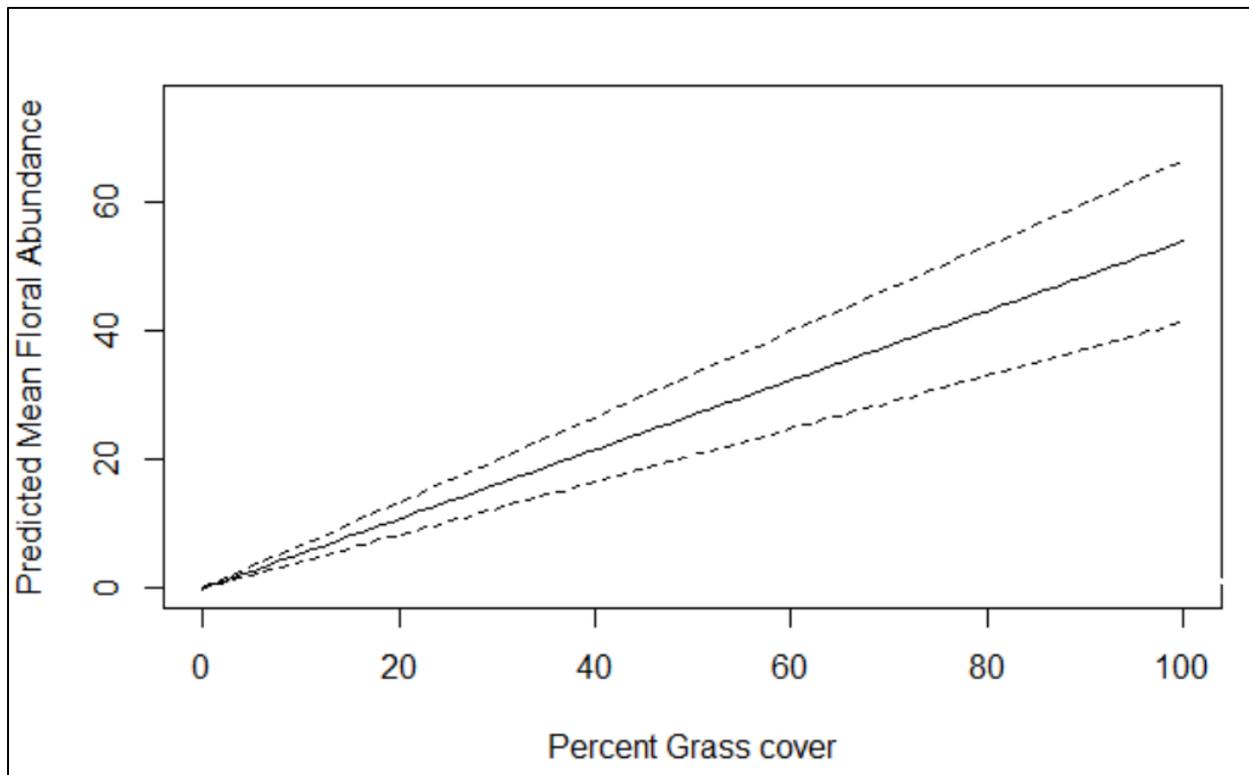
2019 Bee Habitat Associations									
Habitat Variables	V1	V2	V3	V4	V5	V6	V7	V8	V9
Canopy (% cover)		-				-	-		
Large (> 1 m) Saplings (% cover)				-			-		
Small (< 1 m) Saplings (% cover)				-					
Large (> 1 m) Shrubs (% cover)							-	-	-
Small (< 1 m) Shrubs (% cover)							-	-	-
<i>Rubus</i> spp. (% cover)	+	+							+
Forbs (% cover)	+	+		-		+			+
Ferns (% cover)								-	
Grass (% cover)	+	+			+		+	+	+
Coarse Woody Debris (% cover)									
Leaf Litter (% cover)									
Bare Ground (% cover)				-					
Stand Age (# growing seasons)					-	-	-	-	
Elevation (m)				-					
Total Floral Abundance (# flowers)		+		+	+	+	+	+	+
Used Floral Abundance (# flowers)	+	+		+	+	+	+	+	+
Total Floral Diversity (H')	+	+			+	+	+	+	+
Used Floral Diversity (H')	+	+			+	+	+	+	+











**Figure 2.15.** Results of our linear model with grass cover as a positive predictor of mean floral abundance. The dark line shows our model predictions and the dashed lines depict 95% confidence intervals. This model was created with a unique mean floral abundance variable for each site across the entire summer, capturing the variables that affected floral abundance across time.

Notably, floral communities have a dynamic association with stand age (**Fig. 2.16**). Earlier in the growing season (before leaf-out) stand age is either an insignificant or positive predictor of floral abundance. Later in the growing season (after leaf-out), stand age is negatively associated with floral abundance. This likely is because older sites have had more growing seasons post-disturbance for spring ephemerals to establish, but they are not conducive to summer flowering plants. This follows the associations of pollinators and stand age (**Table 2.2 to 2.4**), where we observed that stand age was an insignificant predictor of pollinators in the beginning of the summer but was a consistently negative predictor later in the summer (after leaf-out). This means that younger sites are providing valuable abundant foraging opportunities later in the growing season that do not exist in forest mosaics lacking younger stands.





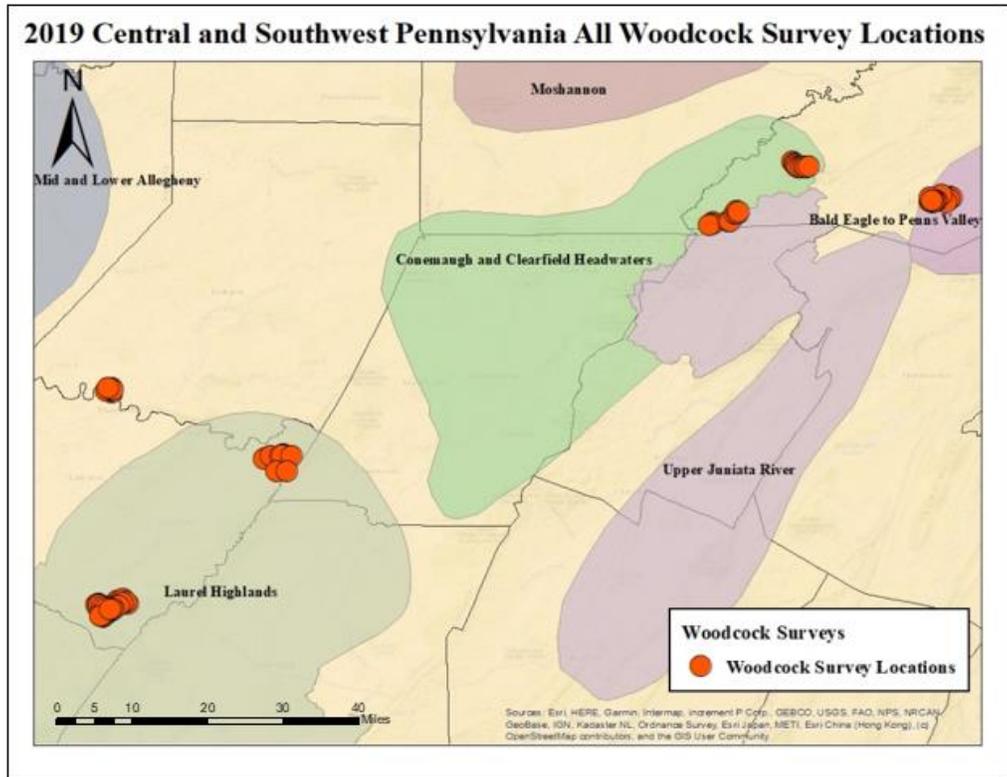


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**Figure 3.1.** Locations of sites in western Pennsylvania where American Woodcock surveys were conducted during April – May 2019. Portions of these sites were managed in recent years to create or enhance early successional communities. Site locations are shown in relation to several focal geographies (Laurel Highlands, Conemaugh and Clearfield headwaters, Upper Juniata River, and Bald Eagle to Penn's Valley) delineated in the R.K. Mellon /National Fish and Wildlife Foundation Western Pennsylvania Business Plan.





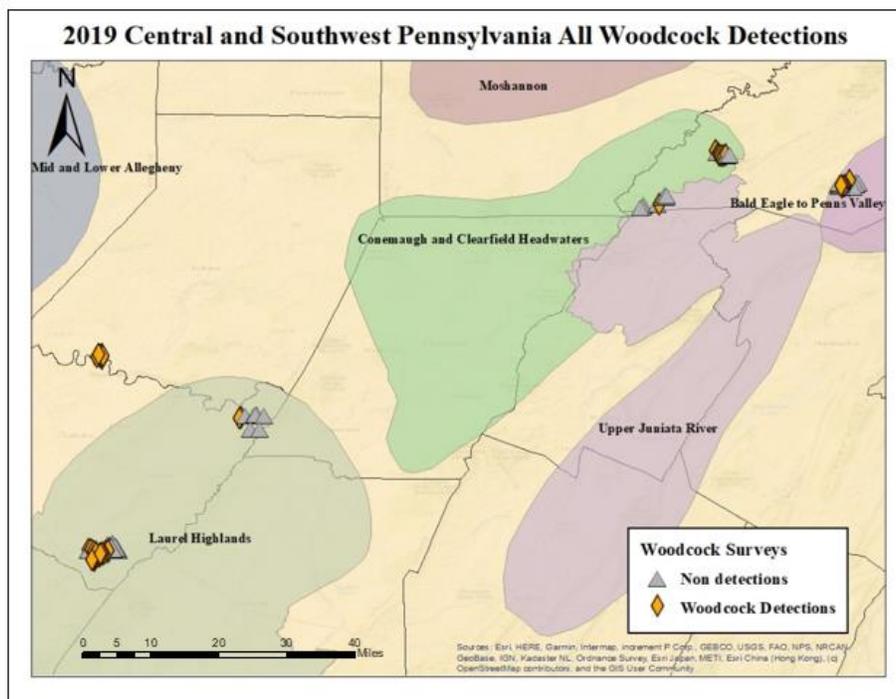
### *Development of Distance-based Thermal Imagery Protocol*

In spring 2019, we used hand-held thermal imaging technology to search for nesting woodcock in managed early successional communities. The goal in 2019, was to pilot the use of the thermal technology to determine its potential use in finding woodcock nests in old field and regenerating timber harvests, and if feasible, develop a distance-based survey protocol to employ in spring 2020. We used two Pulsar Helion XP50 (2.5-20x42mm) thermal imaging monoculars to opportunistically search for nesting females in successional communities and adjacent forest edges. Surveys were conducted in late evenings and early mornings. The location of woodcock we observed were recording using a GPS unit and care was taken not to disturb flush individuals.

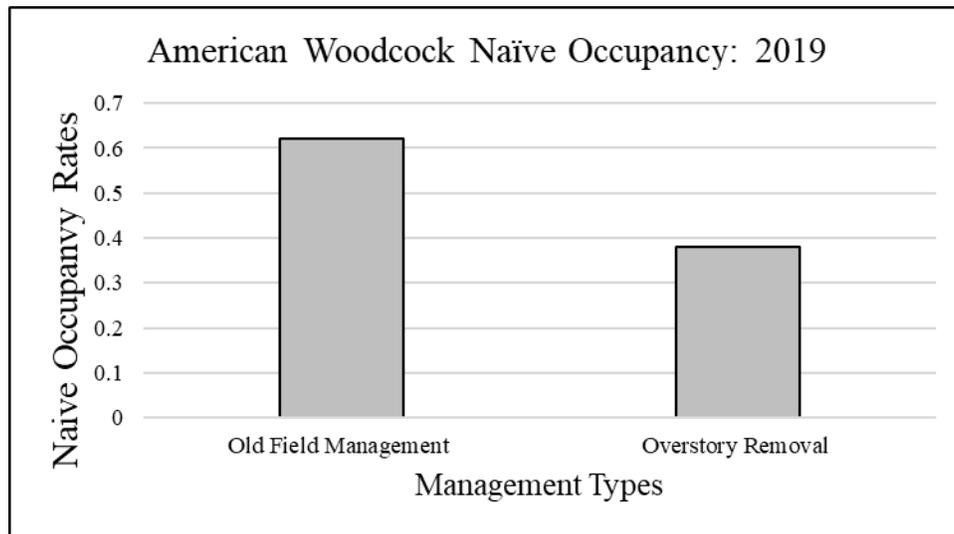
## **Results**

### *Singing Ground Surveys*

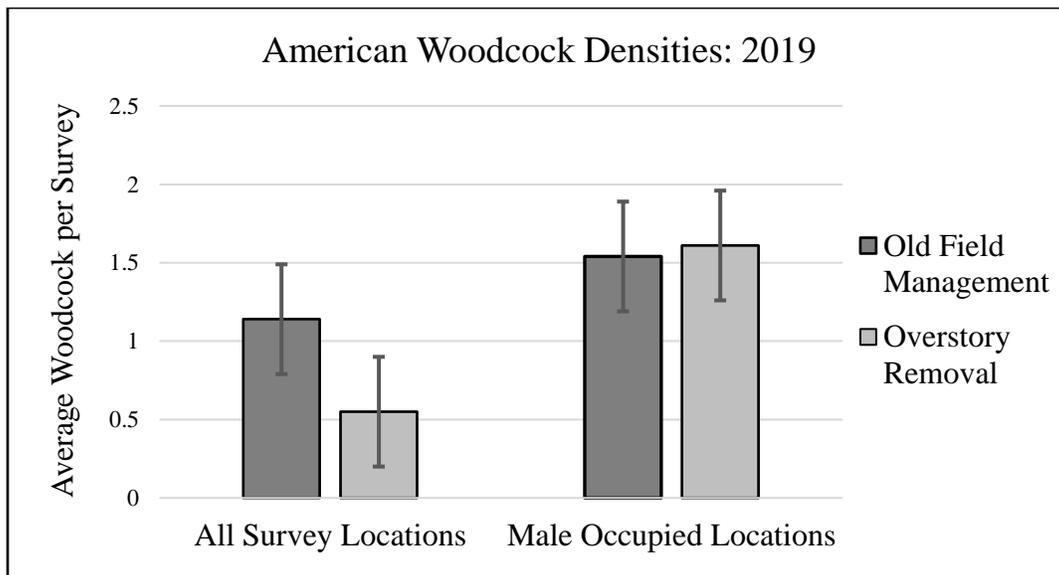
We monitored 88 points across 59 unique early-successional patches: 33 (38%) old field sites and 55 (62%) regenerating timber harvests. American Woodcock (n= 69 males) were detected at 44 of 88 survey points (50% naive occupancy; **Figs. 3.3 & 3.4**). Detections were unevenly distributed among treatment types, with woodcock detected at 24 of 33 (73%) old field management sites and 18 of 55 (33%) regenerating timber harvests. For survey points at which woodcock were detected (n = 42 points), the average number of males detected was 1.57 males/pt (95% CI: 1.32-1.81). The average number of males per occupied point was similar between timber harvests (n = 1.72 males/pt, 94% CI: 1.37-2.07) and old field management (n = 1.58 males/pt, 95% CI: 1.23-1.93; **Fig 3.5**).



**Figure 3.3.** Survey locations in western Pennsylvania where American Woodcock were detected during singing ground surveys conducted between 15 April – 15 May 2019. These sites were managed in recent years to create or enhance early successional communities.



**Figure 3.4.** Naïve occupancy of American Woodcock at managed early successional communities in western Pennsylvania. Surveys were conducted between 15 April to 15 May 2019.



**Figure 3.5.** American Woodcock density for all survey locations and only survey locations where the species was detected. Surveys were conducted in early successional communities located in western Pennsylvania between 15 April – 5 May 2019.

#### *Thermal Imagery Sampling Protocol*

In April 2019, we used hand-held thermal units to opportunistically search for nesting woodcock in managed old fields (n=6) and regenerating timber harvests (n=8). We found that early morning searches were adequate at detecting various species, but, as mornings progressed, solar radiation inhibited the effectiveness of the units. Evening searches were effective at





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## **Part IV: The influence of white-tailed deer (*Odocoileus virginianus*) browsing on vegetation and avian communities in fenced and unfenced timber harvests**

*Prepared by: Halie Parker, Joe Duchamp, and Jeffery L. Larkin, Indiana University of Pennsylvania, PA 15701*

### **Background:**

Deciduous forests of eastern North America have been negatively impacted by white-tailed deer (*Odocoileus virginianus*) browsing over the past several decades (McShea and Rappole 2000; McWilliams et al. 2018). A subcontinental analysis of the United States showed 59% of forest land throughout the Midwest and Northeast exhibited evidence of moderate or high deer impacts (McWilliams et al. 2018). The Mid-Atlantic region maintained the highest proportion (79%) of forest land with moderate or high deer impacts (McWilliams et al. 2018). Chronic browsing by white-tailed deer can negatively affect forest ecosystems by altering plant community structure and species composition (Nuttle et al. 2013; Russell et al. 2001; Tilghman 1989). Specifically, excessive browsing by white-tailed deer can reduce plant growth, recruitment, density, and fitness (Horsley et al. 2003; Rooney & Waller 2003). Additionally, through preferential browsing, deer can shift forest understory composition such that it no longer resembles canopy composition (Long et al. 2007), and chronic and wide-spread browsing that impacts woody species composition can have long-lasting effects (Kain et al. 2011; Nuttle et al. 2011). Abundance of white-tailed deer was found to be just as influential on development of forest understories as stand and site attributes (*e.g.*, aboveground biomass, relative density, and stand age) (Russell et al. 2017).

Excessive browsing by white-tailed deer has been often implicated as a major driver in the failure of forests to successfully regenerate after timber harvest and other disturbances (Apsley & McCarthy 2004; Marquis and Brenneman 1981; Sage et al. 2003). A study in Pennsylvania attributed 50% of forest stand regeneration failures to overbrowsing by deer (Witmer & deCalesta 1991). Regeneration of tree species desired for timber production and wildlife habitat can be substantially reduced by white-tailed deer browsing (Rooney & Waller 2003; Russell et al. 2005), thus lowering future economic and ecological value of forest lands (Marquis 1974; Marquis 1981). Lack of regeneration can cause substantial economic loss, especially to states like Pennsylvania where \$100-500+/ha may be spent to regenerate sites that are negatively impacted by deer (Witmer & deCalesta 1991).

Forest bird communities are especially susceptible to the impacts of deer populations that become out of balance with their local landscapes (Royo & Finley 2013; Russell et al. 2017). This is largely due to the intricate relationship between forest bird communities and vegetation structure (Cody 1981; MacArthur & MacArthur 1961; Mills et al. 1991) as well as relationships between plant species composition and insect prey availability (Bellush et al. 2016; Holmes & Robinson 1981; Robinson & Holmes 1982). Ultimately, deer overbrowsing reduces and degrades understory and mid-canopy vegetation such that it becomes unavailable to forest bird species that require these vegetation strata (Baiser et al. 2008). As such, increases in deer populations in North America have been implicated in the declines of understory-dependent forest-songbird populations (Chollet & Martin 2013). Species that forage or nest near the ground are the most negatively affected guilds (Rushing et al. 2020).

To mitigate deer impacts on forest regeneration, public and private land managers often use deer exclusion fencing until woody stems have become established and grow beyond the reach of deer (1.8m) (Gutstafson 2011; NRCS 2017). Besides harvesting, fencing is often cited as the best means for controlling deer damage (Smith and Coggin 1984). Fences made of woven-wire are erected as long-term barriers to prevent deer from accessing valuable or vulnerable areas (Smith & Coggin 1984; VerCauteren et al. 2006). However, fencing can be expensive, woven-wire fence is \$10-15/m (VerCauteren et al. 2006). Deer exclusion fencing is a conservation practice often employed by public land managers and on private forests enrolled in the Working Lands for Wildlife partnership in Pennsylvania (Gutstafson 2011; NRCS 2017).

We are aware of only two studies that have used deer exclusion fencing to examine the effects of deer browsing on avian abundance and timing of territory settlement (Holt et al. 2010; Holt et al. 2013). These experiments were conducted in Europe and used small (< 0.63 ha) fenced and unfenced control plots. While previous studies provided insight into the effects of deer browsing on vegetation and avian communities, the small size of study plots relative to the scale of operational silviculture may limit transferability of their findings. This limitation is particularly important given the fact that habitat management efforts for forest birds are usually performed at the stand-level scale (*i.e.*, Bakermans et al. 2015; Boves et al. 2013). Indeed, findings from stand-scale studies that use several paired fenced (deer excluded) and unfenced (deer accessible) operational-scale timber harvests that are immediately adjacent to each other are needed to build upon previous studies.

We initiated a study to compare the effect of white-tailed deer browsing between paired fenced and unfenced regenerating timber harvests < 10 years post-harvest in deciduous forests of central Pennsylvania. Specifically, we compared browsing intensity, vegetation density, woody species composition, stem height, and composition of vegetation strata across 10 pairs of fenced and unfenced timber harvests. We also compared avian communities during territory settlement and breeding season. Lastly, we compared territorial male age structure for two focal forest bird species. To our knowledge, no research has directly collected both browsing intensity and vegetation structure and composition simultaneously to examine the direct effects of browsing on vegetation characteristics. We are also unaware of previous studies that used multiple study sites to compare effects of browsing on vegetation inside and outside fenced operational-scale timber harvests. Additionally, to date no research has examined the relationship between deer browsing and timing and selection of territory settlement in forest birds of North America. Ultimately, our study will provide insight regarding the ecological benefits of implementing deer exclusion fencing as a conservation practice to promote healthy and diverse regenerating forests and creating quality wildlife habitat.

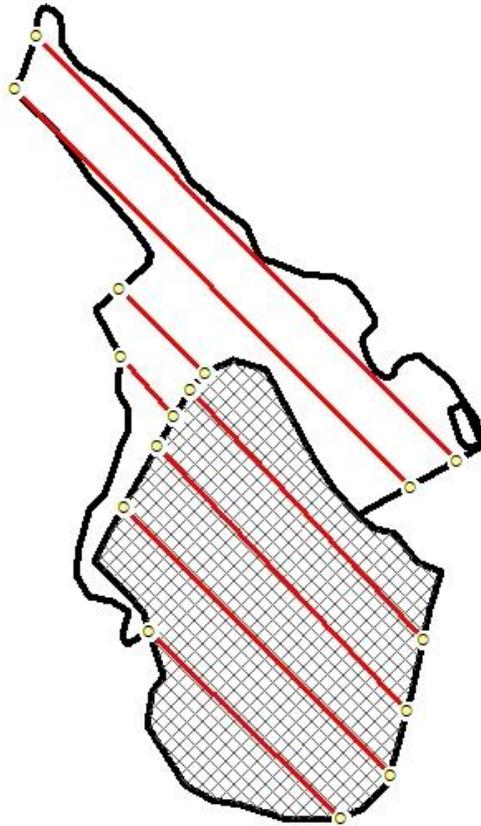
### *Objectives*

1. Compare and describe the effect of white-tailed deer browsing on vegetation structure and habitat characteristics between fenced and unfenced timber harvests
2. Compare the difference in avian communities, density (individuals/ha), and settlement for forest birds between fenced and unfenced timber harvests
3. Compare age structure between fenced and unfenced harvests for two focal forest bird species





plots was dependent on harvest size. We conducted browse intensity surveys from 19 March – 2 April 2019.



**Figure 4.1.** Layout of browsing transects (red lines) for one pair of fenced (hashed; 57ha) and unfenced (open polygon; 44ha) harvests on State Game Lands 033). Our 1m radius sample plots were evenly distributed along each transects and spacing was based on harvest size and transect length.

In each 1m radius plot, we counted, identified, and recorded browse category (unbrowsed or browsed) of all woody stems (saplings and shrubs) and *Rubus* spp. We lumped some species into genera groups if they were difficult to distinguish without leaves such as Oaks (*Quercus* spp.), *Rubus* spp., Birch (*Betula* spp.), and Serviceberry (*Amelanchier* spp.). A stem was counted if it was between 10 cm – 2 m in height. Woody stems >2 m in height were not counted unless they had twigs below 2 m that were accessible to deer (< 1.5 m). We categorized a stem as ‘browsed’ if any portion of the plant exhibited damage to the primary or lateral branches due to deer herbivory. If a 1 m radius plot fell within an area that lacked woody regeneration, we moved the plot to the next closest patch of regeneration in a random direction. The random direction was determined by spinning a compass and we recorded whether a plot needed to be moved. For each woody species, proportions (0-1) were calculated by dividing the number of browsed stems by the total number of stems recorded at each plot and then averaged across plots within either fenced or unfenced areas of each site.

We sampled growing season vegetation characteristics (**Appendix 4.1**) within each timber harvest from 14 June – 22 June 2019. We sampled vegetation at 28 and 27 randomly







distance models are an effective tool for using observed distances to birds to estimate density while accounting for differences in detection probability (Kéry & Royle 2016). We ran our models in the R package *unmarked* (Fiske & Chandler 2011) and compared models based on  $AIC_c$  values. The number of detections for a particular species was summed by point count location and distance bin (0 – 25m, 25-50m, 50-75m, 75-100m). Thus, any detections >100m were not included in this analysis. Also, we excluded detections that were outside of the harvest being surveyed.

When developing our models, we first established a model for detection probability before examining estimates of density. We began by identifying the key function (hazard, exponential or half normal) that best described the relationship between distance and detection probability of each species. We then individually added covariates that may describe additional variability in detection probability. Our detection probability covariates were ordinal date, time of day, wind using the Beaufort wind index (scale of 0-5), cloud cover (0-100%), precipitation (none, fog, mist, light rain, heavy rain, snow), and observer ( $n = 4$ ). All detection covariates that improved the detection model by  $>2.0 \Delta AIC_c$  (Akaike's Information Criterion adjusted for small sample size) (Burnham and Anderson 2003), when compared to the null (intercept-only) model, were incorporated into the models for estimating density.

When examining the variation in density for each focal species, we generated up to 20 models, a null model followed by models containing one of 19 site covariates: age of the harvest (yr), elevation (m), harvest size (ha), canopy cover, >1.5m sapling cover, <1.5m sapling cover, >1.5m shrub cover, <1.5m shrub cover, *Rubus* spp. cover, forb cover, fern cover, grass cover, coarse woody debris cover, leaf litter cover, bare ground cover, *Vaccinium* spp. (blueberry) and *Gaylussacia baccata* (huckleberry) cover, horizontal vegetation density, vertical vegetation density, and treatment (fenced or unfenced). We tested if any variables were strongly correlated (correlation  $\geq 0.7$ ) before running the analysis. We tested each variable independently to determine if they significantly ( $> 2 \Delta AIC_c$  from null model) influenced avian density. All site covariates that improved the density model by  $>2 \Delta AIC_c$ , when compared to the null model, were incorporated into a global model to compare against variables modeled independently. If global modeling revealed a significant variable, we used it to predict the density of individuals in fenced versus unfenced harvests. We used a goodness of fit test to test whether the highest ranked model adequately fit the data (Whitlock & Schluter 2009).

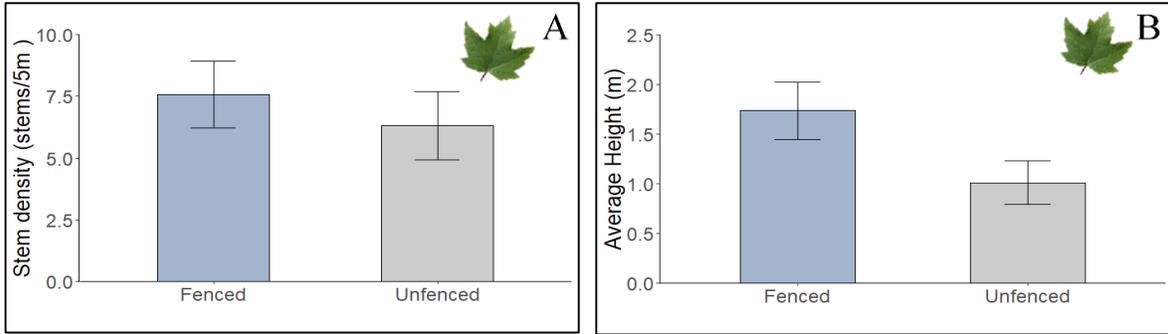
### *Territory settlement*

For the territory settlement analysis of each of our focal species, we compared our observed counts to environmental variables to examine which variables appear most influential for birds when selecting territories. The variables included in these models were horizontal vegetation density, vertical vegetation density, and treatment (fenced or unfenced). For each species, we ran a separate group of models for each of the first five point count visits (*i.e.*, territory settlement visits) (April 11 – May 8).

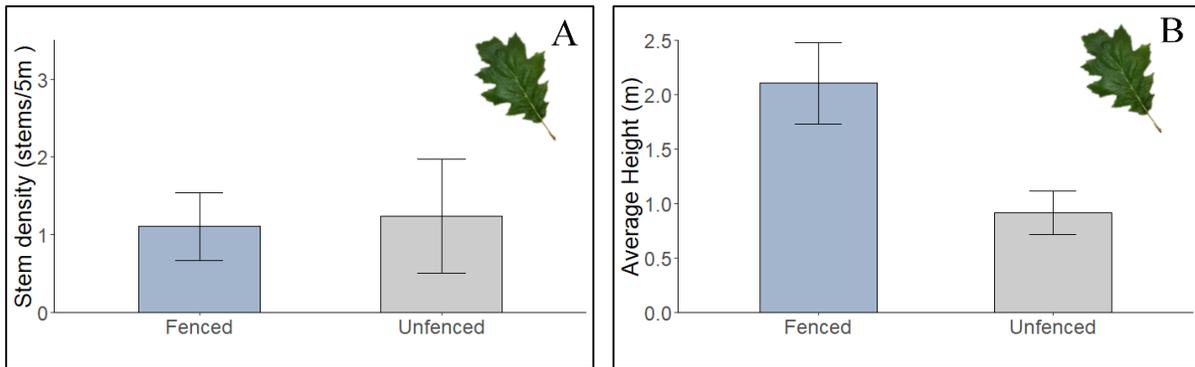
We ran generalized linear models (GLM) using the default Poisson method in R. We built five models for each visit that had detections. These five models included a null model, treatment model, horizontal vegetation model, vertical vegetation model, and a generalized vegetation model, including both horizontal and vertical vegetation. Due to small sample sizes and thus limited power we did not test for multiple  $>2$ ) variables combined. After running each of the five models for each visit, we checked for overdispersion using the generalized vegetation



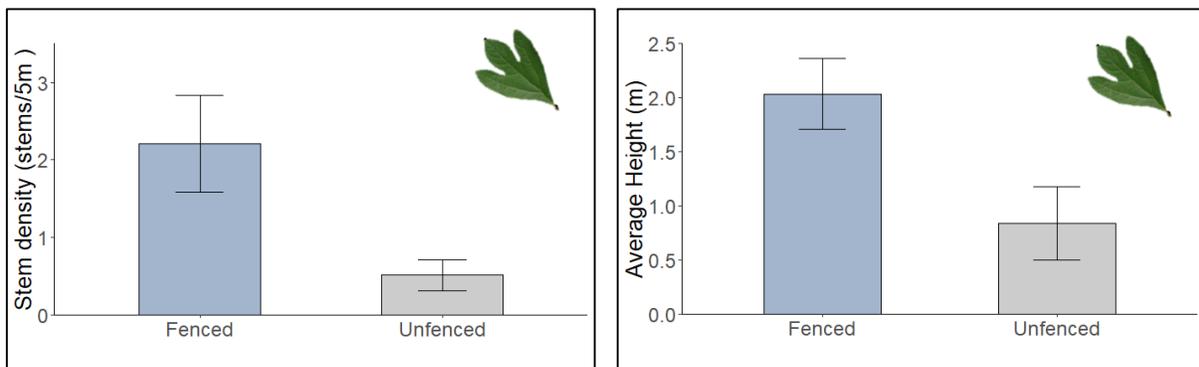




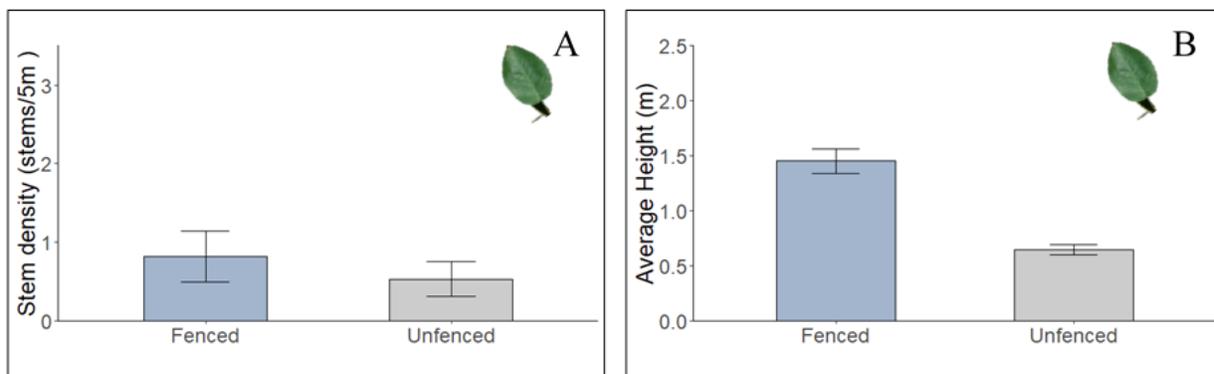
**Figure 4.4.** Average  $\pm$  SEM stem density (stems/5m<sup>2</sup>) (a) and average height (m) (b) for red maple (*Acer rubrum*) in fenced and unfenced harvests.



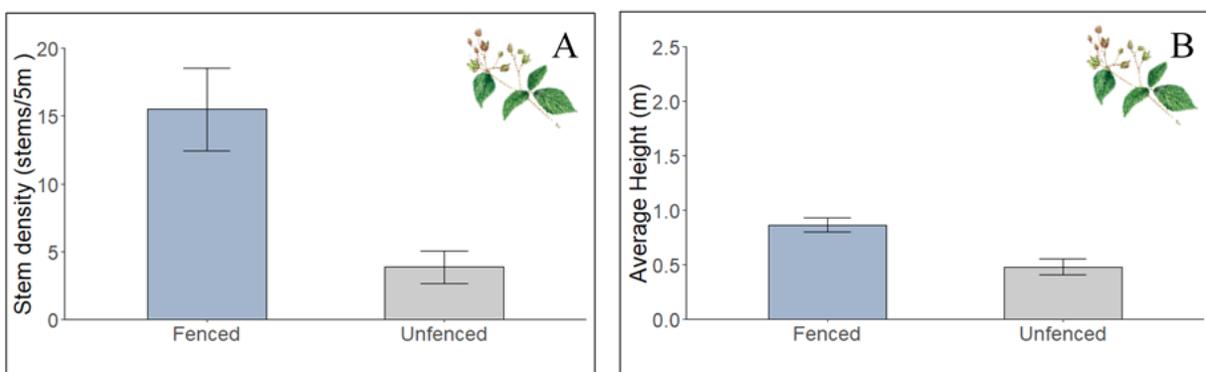
**Figure 4.5.** Average  $\pm$  SEM stem density (stems/5m<sup>2</sup>) (a) and average height (m) (b) for red oak (*Quercus rubra*) in fenced and unfenced harvests.



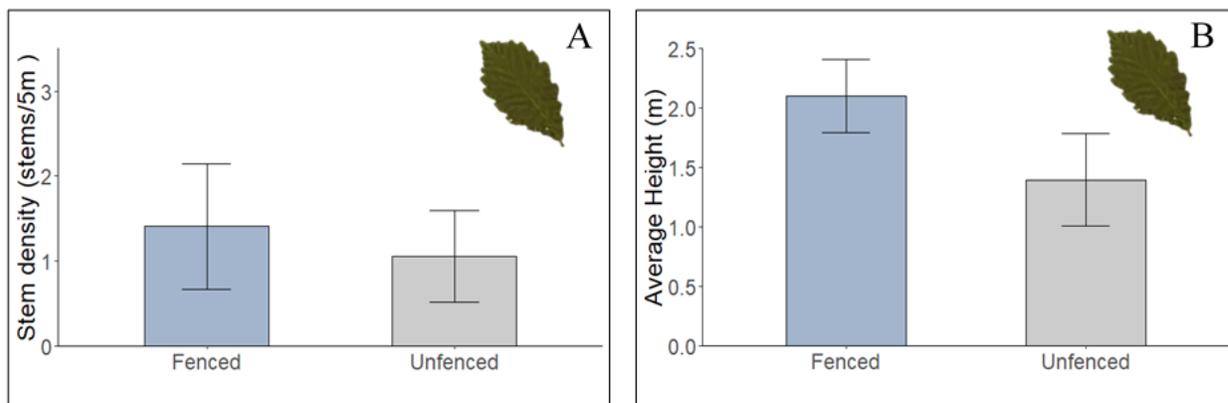
**Figure 4.6.** Average  $\pm$  SEM stem density (stems/5m<sup>2</sup>) (a) and average height (m) (b) for Sassafras (*Sassafras albidum*) in fenced and unfenced harvests.



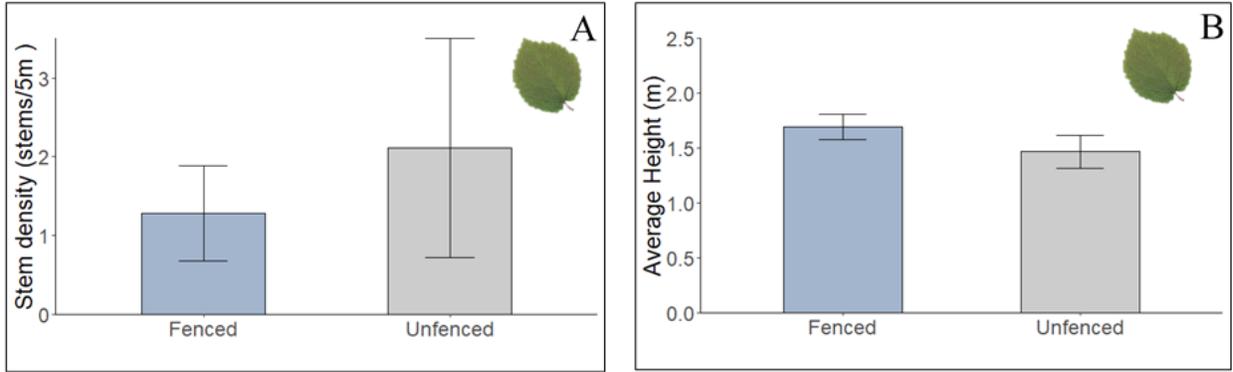
**Figure 4.7.** Average  $\pm$  SEM stem density (stems/5m<sup>2</sup>) (a) and average height (m) (b) for Serviceberry (*Amalanchier* spp.) in fenced and unfenced harvests.



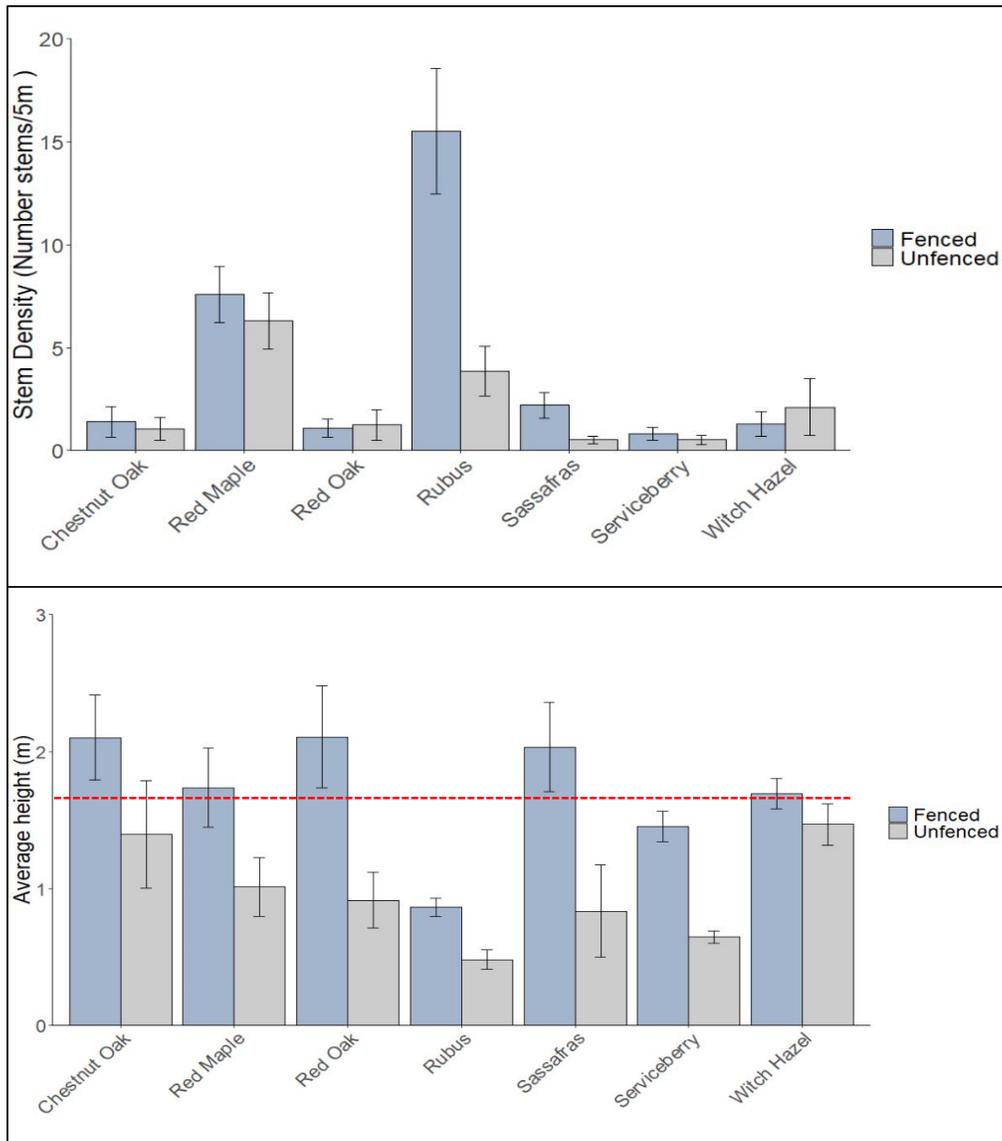
**Figure 4.8.** Average  $\pm$  SEM stem density (stems/5m<sup>2</sup>) (a) and average height (m) (b) for Rubus spp. in fenced and unfenced harvests.



**Figure 4.9.** Average  $\pm$  SEM stem density (stems/5m<sup>2</sup>) (a) and average height (m) (b) for Chestnut oak (*Quercus montana*) in fenced and unfenced harvests.

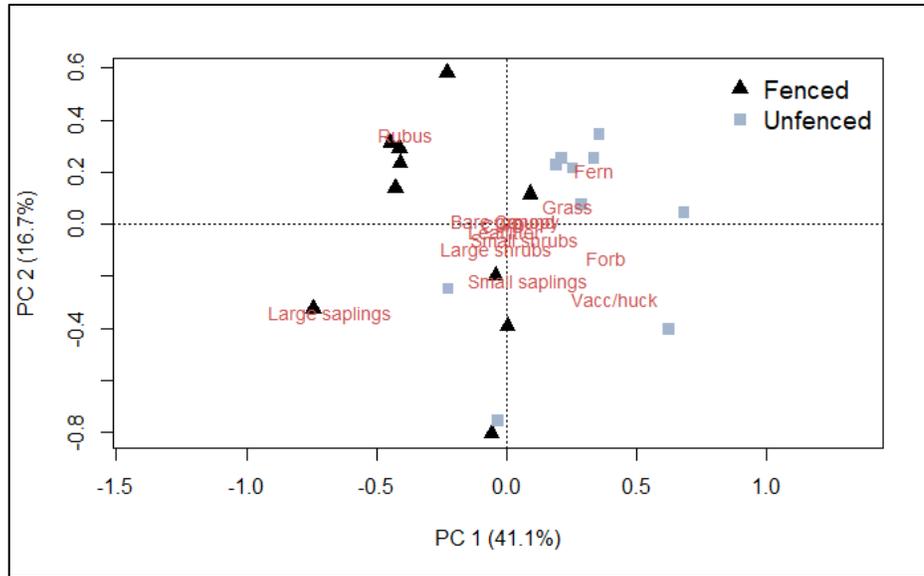


**Figure 4.10.** Average  $\pm$  SEM stem density (stems/5m<sup>2</sup>) (a) and average height (m) (b) for Witch hazel (*Hamamelis virginiana*) in fenced and unfenced harvests.



**Figure 4.11.** Comparison of a) average stem density (stem/5m<sup>2</sup>) and b) height (m) of woody species in fenced and unfenced timber harvests. The red dashed line represents typical height of browse line. Error bars represent standard error of the mean.

The first principal component (PC1) accounted for 41.4% of the variation in vegetation characteristics between fenced and unfenced harvests and was best at distinguishing between fenced and unfenced areas (**Fig. 4.12**). Fenced areas had greater amounts of *Rubus* spp. and >1.5m sapling, while unfenced areas had greater blueberry/huckleberry, forb, and fern (**Fig. 4.12; Table 4.4**).



**Figure 4.12.** Principal components analysis for 13 vegetation cover variables measured in fenced and unfenced timber harvests in central Pennsylvania. Cover variables represent a suite of measures taken using an ocular tube and represent relative amounts of each habitat component.

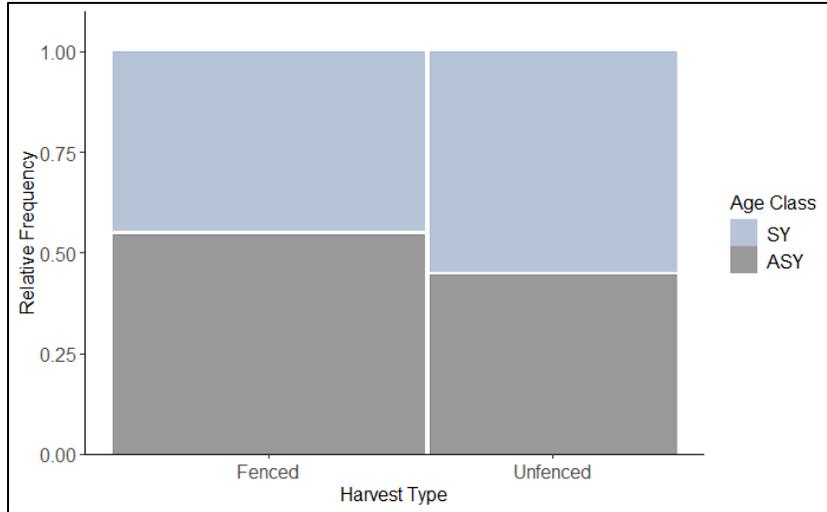
**Table 4.4.** Principal components analysis loadings for primary and secondary components with their percent of variation explained. Cover variables represent a suite of measures taken using an ocular tube and represent relative amounts of each habitat component within paired fenced and unfenced timber harvests.

Variable	PC1 (41.4%)	PC2 (16.7%)
	Loadings	
Canopy	0.08	0.005
Large saplings (>1.5m)	-0.68	-0.35
Small saplings (<1.5m)	0.08	-0.22
Large shrubs (>1.5m)	-0.04	-0.10
Small shrubs (<1.5m)	0.07	-0.06
Rubus spp.	-0.39	0.34
Forb	0.38	-0.13
Fern	0.34	0.21
Grass/sedges	0.24	0.07
Coarse woody debris (CWD)	-0.008	-0.001
Leaf litter	-0.002	-0.02
Bare ground	-0.007	0.004
Blueberry/Huckleberry	0.42	-0.29

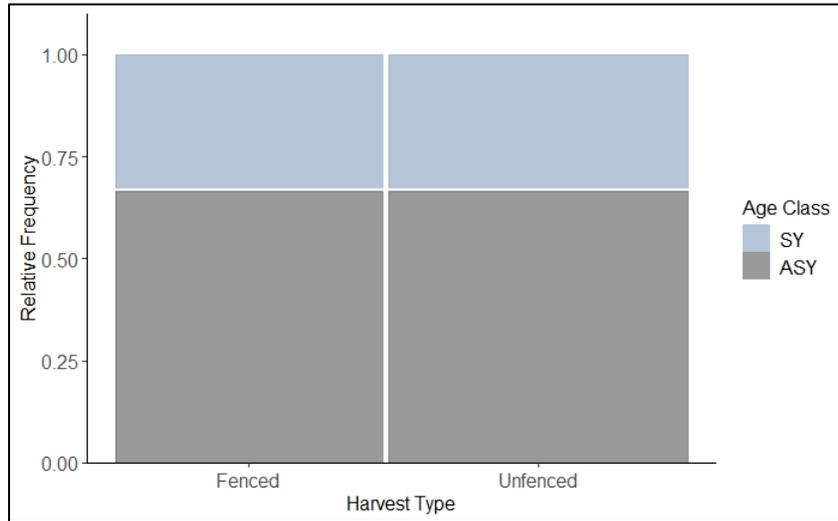








**Figure 4.15.** Chestnut-sided Warbler age class distribution (SY and ASY) by harvest type (Fenced or Unfenced) (n= 156 banded males).



**Figure 4.16.** Common Yellowthroat age class distribution (SY and ASY) by harvest type (Fenced or Unfenced) (n= 147 banded males).

### Conclusions and future efforts

Our browsing results revealed several woody species were significantly more browsed in unfenced versus fenced harvests. While this seems to be a certainty, as those stems in the fence are protected, some species such as Birch, Black cherry, and Serviceberry showed no difference in proportion of stems browsed between fenced and unfenced harvests. This suggests deer may be selectively browsing some species more so than others. It is also particularly noteworthy that 16% of our browse intensity plots in unfenced harvests had to be moved due to lack of regeneration, whereas less than 2% of plots in fenced harvests needed to be moved. Horizontal and vertical vegetation were both significantly higher in fenced harvests, suggesting a more structurally complex understory inside fences. *Rubus* was the only species that significantly differed in stem density (greater inside fences), however five of the seven species we investigated were significantly taller in fenced versus unfenced harvests. Hence, we found









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