

**2019 Annual Report Assessing Wildlife Response to NRCS Conservation Programs
Targeting Eastern Deciduous Forests**

A Conservation Effects Assessment Project (CEAP)

Cooperative agreement # 68-7482-15-501



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Note: *This material is based upon work supported by the Natural Resources Conservation Service, U.S. Department of Agriculture, under number 68-7482-15-501. Any opinions, findings, conclusions, or recommendations expressed in this document are those of the author(s) and do not necessarily reflect the views of the U.S. Department of Agriculture. This is a research progress report that, while representing many of our preliminary findings, is not peer-reviewed and the results contained herein are thus not considered final. This project and its many components are ongoing. Final results and conclusions will be presented in graduate theses/dissertations, NRCS's Conservation Insight series, and peer-reviewed publications.*

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Introduction

Eastern forests of North America and the biodiversity they support are jeopardized by a myriad of threats such as invasive species, diseases, excessive deer browsing, conversion and parcelization, unbalanced age class distributions, lack of natural disturbances (*i.e.*, fire), and unsustainable timber harvest practices (Hain 2006; Rooney & Waller 2003; Dey 2014). Collectively, these factors threaten forest health and resiliency and reduce the population viability of associated species. Indeed, many wildlife species dependent on eastern forests are exhibiting declining populations (*i.e.*, Rosenberg et al. 2019). To reverse these declines, several forest health and habitat restoration efforts have been initiated in recent years by a diverse group of government agencies and NGO conservation groups. While efforts on public forestlands are an important component of these efforts (NABCI 2011), those that focus on private lands are critical given that >70 percent of existing forest cover in the eastern United States is privately-owned (Widmann 2015). The sheer amount of forests under private ownership in the eastern U.S. places a huge stewardship responsibility on the shoulders of private landowners (NABCI 2013).

The U.S. Department of Agriculture's Natural Resource Conservation Service (NRCS) offers financial and technical assistance to private landowners to plan and implement wildlife habitat improvements on their lands (Cuzio et al. 2013). For example, NRCS initiated the *Working Lands for Wildlife* (WLFW) Partnership in 2012 to improve habitat availability for seven imperiled species including the Appalachian population of the Golden-winged Warbler (*Vermivora chrysoptera*). This migratory songbird breeds in heavily forest landscapes of 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 (Roth et al. 2012). Similarly, NRCS initiated an effort to enhance habitat conditions on private forestlands for another imperiled migratory songbird, the Cerulean Warbler (*Setophaga cerulea*), via a project funded through *Regional Conservation Partnership Program* (RCPP). To date, over 17,000 acres have been enrolled in WLFW-Golden-winged Warbler and more than 8,000 acres have been enrolled in the RCPP-Cerulean Warbler.

Certainly, the acres enrolled in NRCS's WLFW and RCPP represent progress toward achieving habitat goals for their respective focal species. However, biological monitoring is also a critical part of evaluating the program outcomes. The results of monitoring wildlife response to programs that aim to restore habitat for focal species like Golden-winged and Cerulean warblers are important for several reasons including: 1) gauging the potential contribution that NRCS programs make toward a species recovery; 2) assessing the need to modify existing habitat management guidelines to improve species-specific outcomes; and 3) quantifying the potential benefits that these programs have for other species associated with these habitats.

In 2015, Indiana University of Pennsylvania and its partners began a collaborative effort to monitor Golden-winged Warbler use of private lands enrolled in NRCS conservation programs and managed habitat on nearby public lands (McNeil et al. *in press*). We also continued a monitoring effort to assess American Woodcock response across many of these same sites. Based on the results of Roberts et al. (2017) and Koh et al. (2016), our partnership initiated a study in 2017 to fine tune a pollinator survey protocol in regenerating timber harvest (McNeil et al. 2018). In 2018 and 2019, we implemented the pollinator survey protocol on over 100 sites managed for Golden-winged Warbler nesting habitat on private and public lands in

Pennsylvania. We also initiated a monitoring program in Pennsylvania (PA) and western Maryland (MD) to assess Cerulean Warbler use of sites enrolled in the RCPP-Cerulean Warbler project. Finally, we conducted the first year of our study evaluating avian and vegetation communities in paired fenced-unfenced regenerating timber harvests. This project is important because deer exclusion fencing is a conservation practice that is often used for WLFW and RCPP projects.

Herein, we present preliminary results for the above-mentioned projects concerning the forest management efforts that target Golden-winged and Cerulean warbler breeding habitat. These include: 1) monitoring Cerulean Warbler and associated species response to forest management on lands enrolled in RCPP in PA and western MD; 2) quantifying pollinator diversity and density within regenerating timber harvests on WLFW lands in Pennsylvania; 3) estimating American Woodcock occupancy and density in early successional communities resulting from timber harvests and old field management; and 4) assessing the value of deer exclusion fencing to avian and plant communities in operational-scale regenerating timber harvests. The monitoring efforts outlined here are essential to ensuring an effective, and ever-evolving, long-term conservation strategy for creating and maintaining breeding season habitat for the Golden-winged and Cerulean warblers, and to understanding the degree to which these efforts benefit associated taxa.

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Part I. Cerulean Warbler and associated species response to forest management implemented through NRCS's Regional Conservation Partnership Program and Working Lands for Wildlife.

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Background

Expansive timber harvesting throughout the eastern United States in the early 1900's resulted in the clearcutting of entire forested landscapes (Williams 1989; MacCleery 2011). While large portions of this region have regenerated as second-growth forest over the past century (Askins 2001), changes in timber harvest practices and lack of other disturbances (*i.e.*, fires) have resulted in millions of acres of even-aged mature forests (*e.g.*, 80 years old) (Shifley et al. 2014). Moreover, it is recognized that these forests are highly uniform in structure and thus do not support the vegetation complexity of pre-European settlement forests (Askins 2001; Shifley et al. 2014). Historically, natural forest disturbances from fires, wind-throw, and tree-senescence occurred regularly, creating canopy gaps of varying sizes allowing early-successional conditions to develop within a matrix of older forest, which resulted in complex vegetation structure (Lorimer 1980). Indeed, the loss of structural complexity in today's eastern forests, driven by limited availability of regenerating young forest (<20 years) and the near absence of old-growth forest (>100 years), have left many taxa that depend on those conditions vulnerable to decline: including bats (Silvis et al. 2016), pollinators (Roberts et al. 2017), and birds (Boves et al. 2013; Schlossberg et al. 2010). Of these taxa, the declines of forest birds have been well-documented (Rosenberg et al. 2019).

The Cerulean Warbler (*Setophaga cerulea*) is a songbird exhibiting one of the most dramatic and sustained population declines of all eastern forest birds. Within the eastern forest region, which host >70% of the species' breeding population (Rosenburg et al. 2016), the Cerulean Warbler populations have declined 2.8% annually since 1966 (Sauer et al. 2017). This canopy-nesting species is associated with habitat conditions characteristic of late-successional deciduous forests including structurally complex canopies with frequent gaps (Boves et al. 2013a; Wood et al. 2013). Because late-successional forest conditions are rare in the today's eastern forests, the Cerulean Warbler is predicted to decline by an additional 50% in the next 25 years (Rosenburg et al. 2016). However, studies have shown that forest management can be used to create the structural conditions that Cerulean Warbler require (*i.e.*, Boves et al. 2013b). Over the past two decades, research has focused on Cerulean Warbler breeding season ecology with respect to silviculture treatments (*e.g.*, timber harvests) with a goal of developing species-specific habitat management guidelines (Wood et al. 2013; Raybuck et al. 2020). Experimental forest harvests found that basal area (RBA) between 40-90 ft²/acre supported the largest increase in Cerulean Warbler density (Wood et al. 2013). Moreover, a study in Pennsylvania found habitat use by Cerulean Warbler during the post-fledging period was characterized by dynamic use of structural conditions. The implementation of science-based habitat guidelines for Cerulean Warbler is a conservation priority (Nareff et al. 2019).

In 2015, the Appalachian Mountain Joint Venture (AMJV), American Bird Conservancy (ABC), and its many partners initiated a project through the Natural Resource Conservation

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.

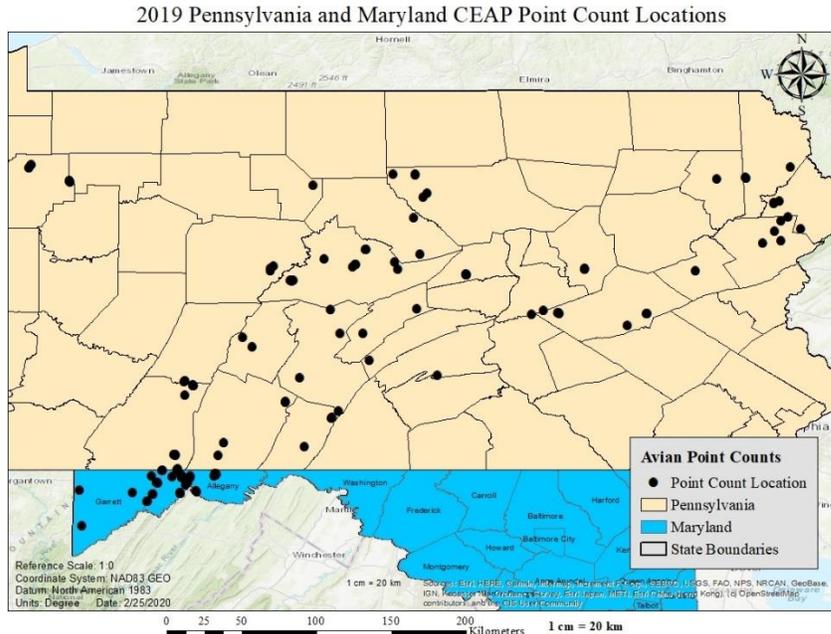


Figure 1.1. Locations of avian surveys within managed and unmanaged forests in Pennsylvania and western Maryland during May-June 2017-19. Note: Dots are offset random distances from true locations to protect landowner privacy.

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

competition); “2” (dominant, trees with crowns extending above the general level and receiving full sun); “3” (co-dominant, trees forming part of the general level and receiving full light from above but not sides); “4” (intermediate, crowns either below or extending into the canopy formed by the dominant and codominant trees with low light from below); and “5” (overtopped, trees completely below the general level receiving no direct light).

Analyses

We treated each year and site as independent random samples, allowing us to pool data from all points monitored over the four-year period. Additionally, we grouped the points into 4 categories based on management status and time since management. The four categories were a) unmanaged points; b) shelterwood harvests with no growing season post-management (0 GS); c) shelterwood harvests with one growing season post-management (1 GS); and shelterwood harvests with more than one growing season post-management (>1 GS).

We used Program R (R Core Team 2019) to run a one-way Analysis of Variance (ANOVA) and Tukey post-hoc test ($p \leq 0.05$) to assess whether avian communities (# individuals/point, species richness (S)/point, and effective species unit/point) differed among the four management categories. We used a two-sample t-test ($p \leq 0.05$) to compare avian communities (# individuals/point, species richness (S)/point, and effective species unit/point) for a subset of points for which we had pre- and post-treatment data. We assessed individual species response to management by comparing naïve occupancy among the four treatment categories using 2019 avian and vegetation data. For this preliminary analysis, we used >10% change in a species naïve occupancy infer differences. We used the package *treemapify* in program R to create treemap diagrams to visualize the relative abundance of each species within each management category. To examine if vegetation metrics differed among management categories, we calculated averages and compared 95% confidence intervals. We considered non-overlapping 95% confidence intervals to be indicative of significant differences.

Results

From 2016 to 2019, we monitored a total of 127 unique points within privately-owned forests. The number of points we surveyed increased annually as the number of landowners enrolled in the RCPP program increased (2016, n=2 pts; 2017, n=12 pts; 2018, n= 64 points; and 2019, n=121 points). A total of 398 point count surveys were conducted during the 4 years. These 398 surveys represented four management categories; unmanaged (pre-shelterwood) (n=190); shelterwood with no growing season post-management (0 GS; n=118); shelterwood with one growing season post-management (1 GS; n=56); and shelterwood with more than one growing season post-management (>1 GS; n=34). For all analyses, we used the higher number of individuals of a species detected during the two surveys conducted annually at each point.

Avian Communities

Over the 4 years of surveys, we detected a total of 3,556 individual birds representing 75 species across all management categories. Sites with 1 GS and >1 GS had higher avian abundance (**Fig. 1.2A**; ANOVA, $F=5.36$, $p =0.001$), species richness (**Fig. 1.2B**; ANOVA, $F=5.47$, $p =0.001$), and effective species units (**Fig. 1.2C**; ANOVA, $F=5.16$, $p=0.002$) compared to unmanaged and managed with 0 GS sites.

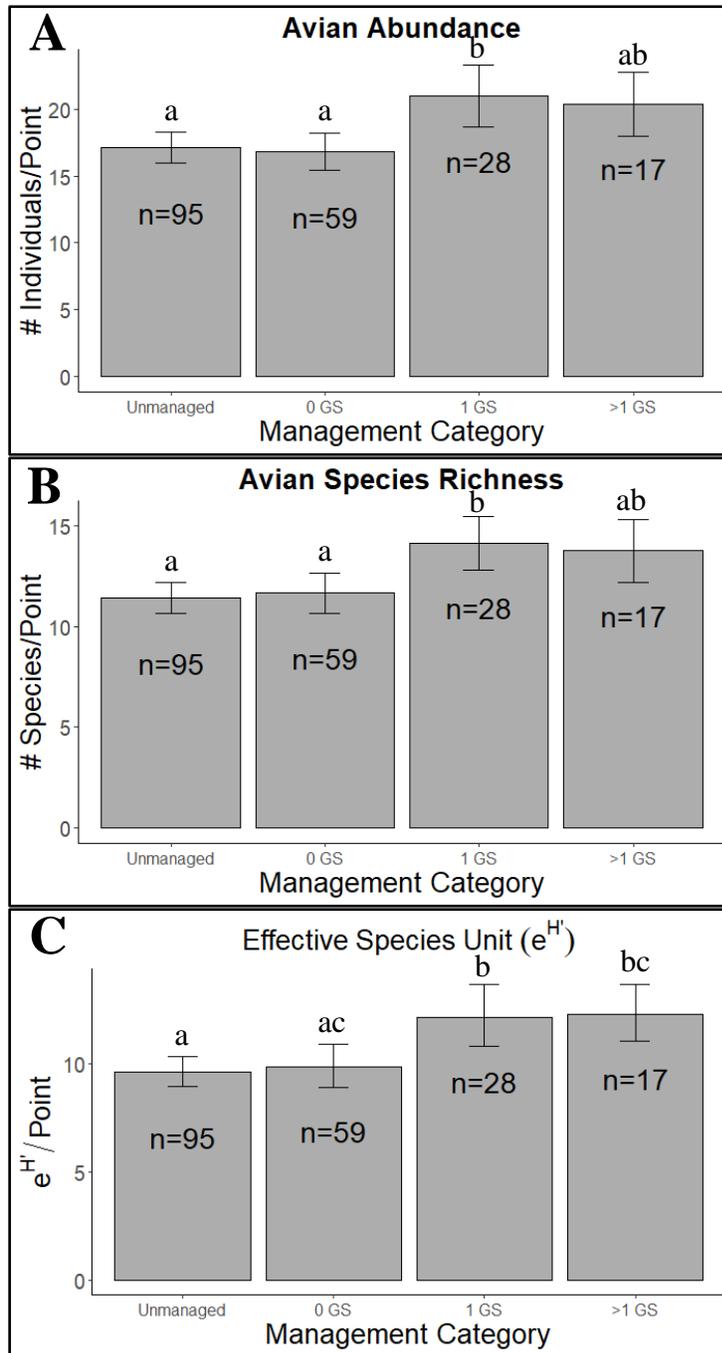


Figure 1.2. Mean avian abundance (A), mean avian species richness (B), and effective species unit (C) for managed and unmanaged forests in western Maryland and Pennsylvania. Surveys ($n=398$) occurred from May-June 2016 across 127 unique point count locations. Bars represent 95% confidence intervals. 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>

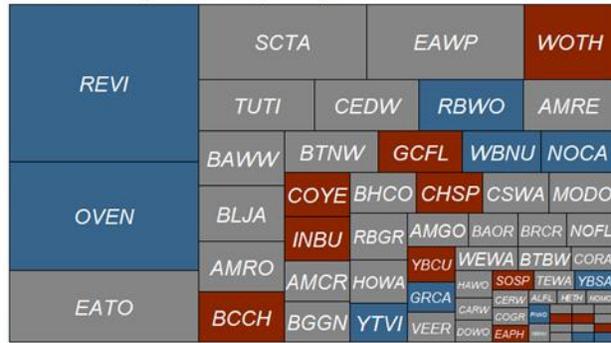
Species	Unmanaged (n=95)			0 GS (n=59)			1 GS (n=28)			>1 GS (n=17)		
	Points	Ind.	Naïve Occ. (%)	Points	Ind.	Naïve Occ. (%)	Points	Ind.	Naïve Occ. (%)	Points	Ind.	Naïve Occ. (%)
BAWW	31	37	32.6	21	23	35.6	11	17	39.3	8	9	47.1*
BLBW	7	8	7.4%	No Detections			3	4	10.7	4	6	23.5*
BRCR	11	11	11.6*	6	8	10.2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

Table 1.2. Avian species that experienced a significant change (>10% change) in naïve occupancy (percent of survey points a species was detected) among management categories (Unmanaged, 0 growing seasons (GS), 1 GS, and ≥1 GS). The table includes the number of individuals detected, number of points detected at, and naïve occupancy of for each species in each management category). Asterisks (*) denotes highest naïve occupancy. Species common names and the 4-letter banding code can be found in Appendix 1.1.

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. (%)
AMRE	27	47	28.4	16	23	27.1	11	15	39.3	8	8	47.1*
AMRO	26	33	27.4	14	21	23.7	6	6	21.4	6	8	35.3*
BAWW	31	37	32.6	21	23	35.6	11	17	39.3	8	9	47.1*
BCCH	19	22	20.0	13	21	22.0	3	4	10.7	4	5	23.5*
BGGN	15	17	15.8	10	13	16.9	8	9	28.6*	3	4	17.6
BHCO	13	14	13.7	11	13	18.6	11	11	39.3*	2	2	11.8
BLJA	41	57	43.2*	18	23	30.5	8	9	28.6	6	6	35.3
CEDW	36	47	37.9	21	26	35.6	13	19	46.4	8	10	47.1*
COYE	13	22	13.7	10	14	16.9	9	13	32.1	8	12	47.1*
CSWA	16	19	16.8	11	13	18.6	10	15	35.7*	5	7	29.4
EATO	33	54	34.7	37	66	62.7	19	36	67.9	14	24	82.4*
EAWP	60	80	63.2	46	57	78.0*	20	27	71.4	12	21	70.6
GCFL	15	17	15.8	14	17	23.7	7	12	25*	2	2	11.8
GRCA	16	18	16.8	7	7	11.9	2	2	7.1	5	5	29.4*
HOWA	13	19	13.7	9	12	15.3	7	13	25.0*	4	7	23.5
INBU	17	20	17.9	13	14	22.0	13	21	46.4*	3	4	17.6
OVEN	85	207	89.5	46	100	78.0	20	55	71.4	16	45	94.1*
RBGR	17	18	17.9	10	13	16.9	9	9	32.1	8	10	47.1*
RBWO	33	36	34.7	23	26	39.0	15	15	53.6*	5	5	29.4
REVI	94	263	98.9*	57	144	96.6	26	77	92.9	14	35	82.4
SCTA	68	96	71.6	45	61	76.3	20	28	71.4	14	18	82.4*
TUTI	39	46	41.1	23	29	39.0	14	16	50.0*	5	7	29.4
WBNU	22	23	23.2	15	16	25.4	9	9	32.1	8	8	47.1*
WEWA	17	19	17.9	6	7	10.2	9	11	32.1*	1	1	5.9
WOTH	47	85	49.5	21	33	35.6	15	20	53.6*	3	5	17.6

Vegetation Surveys

Average basal area was lower at managed sites with 0 GS (basal= 63.5ft²/ac; 95%CI: 57.6-69.3), 1 GS (basal=57.9 ft²/ac; 95%CI: 50.0-65.8) and > 1GS (basal=72.6 ft²/ac; 95%CI: 63.5-81.8) than unmanaged points (91.8 ft²/ac; 95%CI: 87.4-96.2). Percent cover of several vegetation strata also differed between unmanaged and managed sites whereby canopy, leaf litter, and >1m tall sapling cover were lower at managed sites compared to unmanaged sites (**Fig. 1.4**). Points with 1 GS and >1GS had more <1m tall sapling, forb, and grass cover compared to unmanaged and 0 GS points. When combining all three managed types (0, 1, >1 GS), average tree DBH was higher in managed (33.6 cm, 95%CI: 33.0-34.2) than unmanaged (31.4 cm, 95%CI: 30.9-31.9) sites. Average tree DBH for tree species preferred by Cerulean Warblers did not differ between managed (33.9 cm, 95%CI: 33.2-34.6) and unmanaged (32.5 cm, 95%CI: 31.7-33.2) sites. When combining all managed types (0, 1, >1 GS), the proportion of trees that were species preferred by Cerulean Warblers was higher for managed (0.49, 95%CI:0.44-0.54) than unmanaged (0.37, 95%CI: 0.33-0.42) sites.

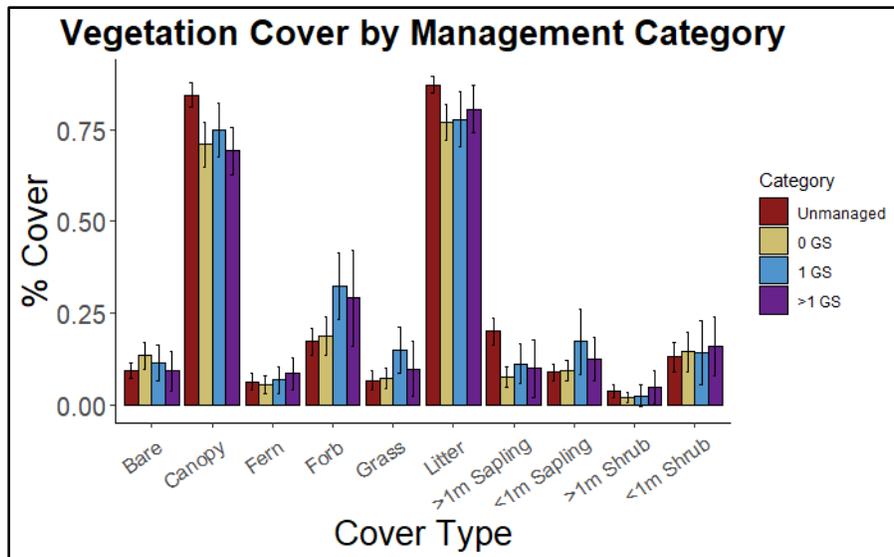


Figure 1.4. Percent cover for vegetation strata in four forest management categories (unmanaged, 0 GS, 1 GS, and >1 GS) in Pennsylvania and western Maryland. Data were collected at avian survey location in June 2019. Note: GS= Growing Seasons, Bars represent 95% CIs.

Pre and post-management sites

Of the 127 unique point count locations we surveyed, 18 were locations for which we collected pre-management (unmanaged) and post-management (0 GS) avian and vegetation data. We observed a decrease in avian abundance (# individuals/point) following management (t-test, $t=-2.68$, $P=0.01$) and no change in species richness (t-test= -1.76 , $P=0.09$) or effective species unit (e^H) following management (t-test, $t=-1.71$, $p=0.10$) (Fig. 1.5). We detected no change in Cerulean Warbler occupancy across these 18 sites pre- and post- management (1 of 18 occupied). Average basal area was higher pre-management ($76.7 \text{ ft}^2/\text{acre}$; 95% CI: 68.5-76.7) compared to post-management (0GS; $59.3 \text{ ft}^2/\text{acre}$; 95% CI: 51.2-66.8). We only measured DBH at these sites post-management, and the average DBH of all trees and preferred trees were 35.9 cm (95% CI: 34.5-37.2) and 36.6 cm (95% CI 34.6-38.6), respectively. The proportion of trees that were preferred by Cerulean Warblers did not differ between pre-(0.42; 95% CI: 0.32-0.53) and post-(0.35; 95% CI: 0.27-0.43) management. No percent cover estimates differed between points pre- and post- management. As we increase our sample size for sites with pre and post-management data, we will be able to better assess avian and vegetation response to management.

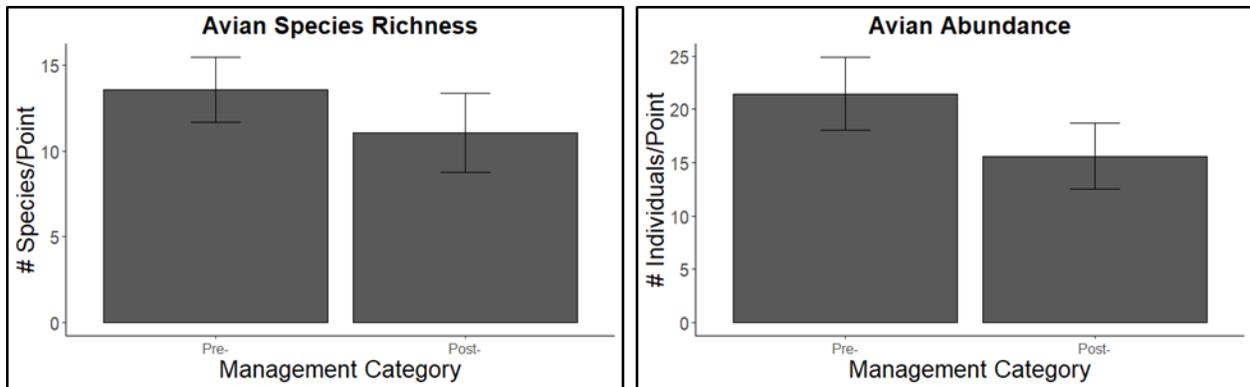


Figure 1.5. Mean avian species richness/point (a) and mean avian abundance (individuals/point; b) at points monitored both pre- and post- forest management. Surveys occurred from May-June 2016 to 2019 at 18 point count locations in Pennsylvania and western Maryland. Note: Bars represent 95% confidence intervals.

Conclusions and Future Efforts

The primary management practice evaluated in our study was shelterwood timber harvest, which reduces basal area by removing undesired tree and sapling species to create gaps in the canopy that promote understory regeneration. Cerulean Warbler response (occupancy and abundance) to shelterwood practices can occur soon after treatments are complete (<1 growing season) (Boves et al. 2013). Average basal area in the managed sites we monitored ranged from 62-73 ft²/acre fell within the range for optimal residual basal area (40 - 90 ft²/ac) listed in the Cerulean Warbler habitat guidelines (Wood et al. 2013). In fact, the average basal for our unmanaged sites was 91 ft²/acre, which also falls close to the range recommended values and is certainly much less than found in unmanaged mature eastern deciduous forests (>120 ft²/acre). Nonetheless, Cerulean Warbler naïve occupancy was low (0.16) across all managed sites we monitored.

With basal values for all our sites so closely in line with those recommended for Cerulean Warblers, why were so few individuals observed? The answer likely has to do with the DBH values of these forests. Tree size is known to strongly influence Cerulean Warblers occurrence, with the species preferring sites hosting dominant trees >40 cm (16 inches) DBH with broad crowns (Boves et al. 2013a; Wood et al. 2013). However, average tree DBH for our study sites was only 33.6 cm (~13 inches) in managed and 31.4 cm (12 inches) in unmanaged sites. These values fall far below the recommended 40 cm DBH threshold. Given the low DBH values we observed across all sites, it is not surprising that so few Cerulean Warblers were detected at managed and unmanaged sites (naïve occupancy = 0.11; 14/127 points). The fact that the unmanaged forests we monitored had reduced basal area and both managed and unmanaged forests had rather small diameter trees is concerning. Such characteristics are often indicative of past forest management that involved high grading, an unsustainable harvest approach.

Cerulean Warblers have also shown a strong preference for certain tree species. White oak (*Quercus alba*), chestnut oak (*Q. montana*), sugar maple (*Acer saccharum*), cucumber magnolia (*Magnolia acuminata*), and hickories (*Carya* spp.) are preferred by Cerulean Warblers,

whereas black oak (*Q. velutina*), northern red oak (*Q. rubra*), scarlet oak (*Q. coccinea*), and red maple (*Acer rubra*) are avoided (Boves et al. 2013a). As such, management for Cerulean Warblers should seek to retain large individuals of preferred tree species while removing competing/undesired species when possible to achieve reduced basal area targets (Wood et al. 2013). Our vegetation surveys reveal that implementation of management is helping to bring the tree species composition in line with Cerulean Warbler habitat guidelines, as managed sites had a higher proportion preferred tree species. Specifically, when combining data from all managed types (0 GS, 1 GS, >1 GS), the proportion of trees that were species preferred by Cerulean Warblers was higher for managed (0.49) than unmanaged (0.38) sites. Ultimately, the management actions implemented in these forests should be considered rehabilitative and appear to have placed these sites on a trajectory to be more resilient and future Cerulean Warbler habitat once residual trees grow.

While Cerulean Warblers did not exhibit an immediate response to forest management, many other species did. Indeed, forest management that targets Cerulean Warblers can also benefit other disturbance-dependent species and may also retain closed-canopy species but at reduced levels (Sheehan et al. 2013). Our findings support this general pattern of avian community response to forest management. Our monitoring efforts recorded 75 species across all sites monitored, of which 16 species are listed as “species of greatest conservation need” (SGCN) by the Pennsylvania Wildlife Action Plan. Species-specific response to management was mixed with some species showing higher naïve occupancy in managed sites (*e.g.*, Rose-breasted Grosbeak, Eastern Wood-Pewee, Eastern Towhee, American Redstarts), while others exhibited lower naïve occupancy post-management (*i.e.*, Red-eyed Vireo). With this said, most species were impartial or showed minor reductions in naïve occupancy. Interestingly, the Ovenbird exhibited lower occupancy in managed sites immediately following management (0 GS and 1GS), but for managed sites with >1GS the occupancy was at a level similar to unmanaged forest. In the end, landscape scale forest management that strives to ensure the availability of diverse harvest intensities and older unharvested forest will best meet the full-breeding season needs of a diverse forest bird community.

In 2020, we will conduct monitoring across additional sites enrolled in the RCPP. As we continue to monitor these points and increase our sample size, we will improve our ability to discern what may be driving Cerulean Warbler occupancy, associated species abundances, and community composition. We will also develop more complex analyses and models to relate individual species abundance/ occupancy and community composition to vegetation data. Additionally, we will quantify other factors known to be important to Cerulean Warblers and include them in our analyses. These include proximity to known breeding populations and topographic position (aspect) (Wood et al. 2013). Finally, we intend to employ automated recording unit technology (ARU) at each point count location. The use of ARUs will result in a more thorough assessment of species occupancy due to increased sampling effort and will likely improve our detection of uncommon species such as Cerulean Warbler (Bobay et al 2018). After the 2020 field season, we will process ARU data and develop dynamic occupancy models for several focal species. The occupancy modeling will incorporate within-stand and landscape scale metrics as covariates.

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Part II. Habitat Ecology of Native Pollinators in forests managed through NRCS's *Working Lands for Wildlife Partnership* and similar efforts on public lands in the Central Appalachian Mountains

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Background

Pollinators (*e.g.*, bees, butterflies) are functionally important to nearly all terrestrial ecosystems, providing essential pollination services to most 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; Milam et al. 2018).

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 (*Vermivora chrysoptera*; GWWA; USDA 2014; McNeil et al. 2017). Moreover, the fact that these habitats have already demonstrated to provide habitat for many species (McNeil et al. 2018, 2019) further supports the idea that restoration of young forest communities may also provide benefits to insect pollinators. Considering 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 communities. In this study, we examined the ecology of native pollinator populations within early successional forests created through silvicultural practices on public and private lands in the central Appalachian Mountains of Pennsylvania.

Objectives

The overall objective of this CEAP component is to determine habitat characteristics within early successional forests of Pennsylvania that can be managed for to promote native pollinator density and diversity. Specific project objectives include:

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 the abundance and diversity of the floral and pollinator communities within recently harvested early successional forests of the central Appalachian region.
3. Investigate whether stand age, elevation, patch-size, and floral resource availability drive variation in pollinator abundance and diversity within recently harvested early successional forests of the central Appalachian region.

Methods

Study Area

To investigate native pollinator populations through Pennsylvania, we conducted surveys in 2018 (n=75 sites) and 2019 (n=100 sites) high-elevation portions of the state. Our sites included private (2018: n=38; 2019: n=46) and public (2018: n=37; 2019: n=54) lands managed by regeneration silviculture (Fig. 2.1). All sites were recently (<10 years) managed through overstory removal (regeneration) harvests, with our second year of surveys focusing on younger sites (≤ 6 years). Management of private lands followed conservation plans associated with the USDA-Natural Resource Conservation Services (NRCS) WLFW partnership. 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 outdoor sporting organizations. We selected public lands that were near 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, McNeil et al. 2019).

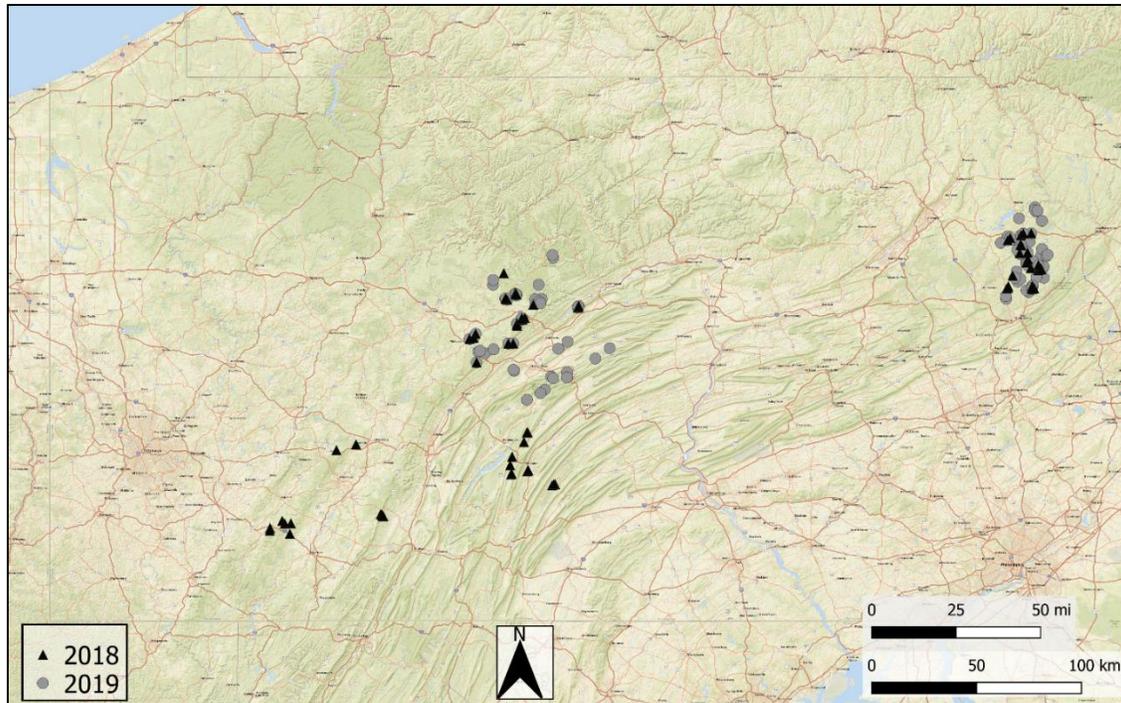


Figure 2.1. Map showing the overstory removal harvests where we surveyed for native bee communities, floral communities, and associated structural habitat characteristics. Sites surveyed in 2018 are represented with a black triangle and sites surveyed in 2019 are represented with a grey circle. Some 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).

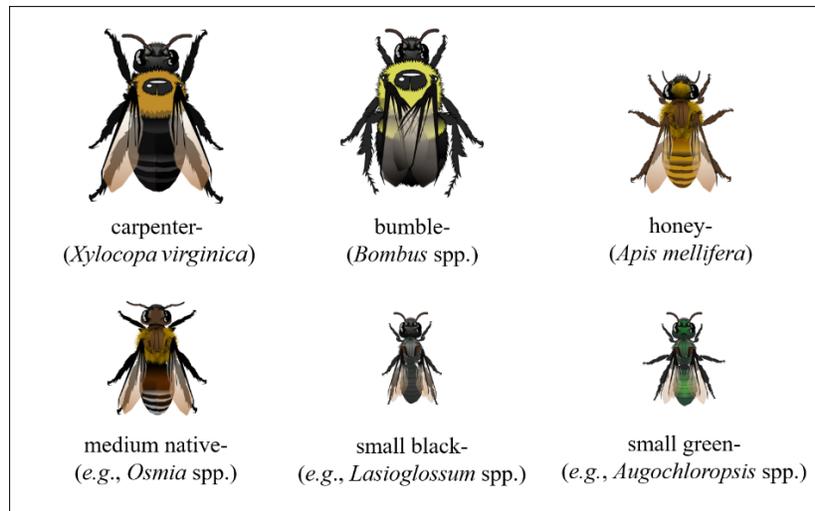


Figure 2.2. Due to the difficulty of identifying bees to species when they are flying, we identified bees to six ‘morphospecies’ groups based on body size and color.

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).

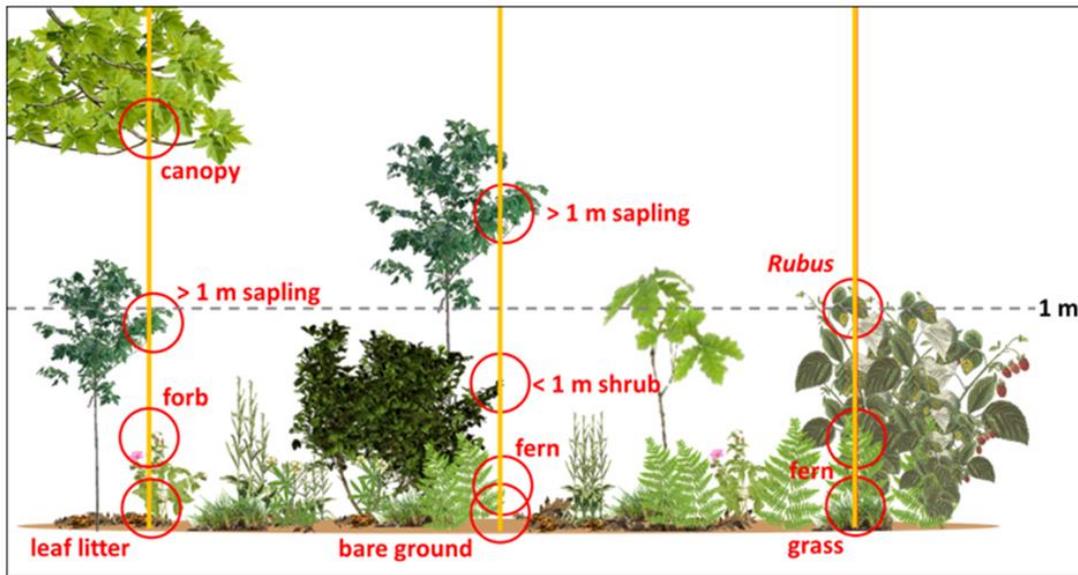


Figure 2.4. An example of the vegetative survey conducted. Pictured are three vegetation profiles that would be measured using an ocular tube and placed 10 m apart on the transect line. For each tube reading (yellow vertical line), the strata considered 'present' are circled and noted in red.

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.

Pollinator Density

We analyzed our data in program R (R Core Team 2019) with the packages *unmarked*, *AICmodavg*, *vegan*, and *dplyr*. Using methods defined by McNeil et al. (2019) and Kery and Royle (2015), we estimated 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 (morphospecies combined) and all butterflies (all species combined). Each model contained a single habitat covariate of one of the following: canopy cover, 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 constructed two sets of variables for our floral community: total floral availability and used floral availability. Used floral availability metrics were calculated from a subset of our floral survey data that was determined to be preferentially selected for by pollinators on these sites through our visual survey results. We calculated the floral diversity at each site using the Shannon-Wiener Diversity Index (H') (Jost 2006). We ran hierarchical distance 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 & Anderson 2003).

Pollinator Diversity

We calculated a diversity value for each site for the combined trap data (effective species unit, Jost 2006). Because the weather for each trap-day varied, we used nearby weather station data to calculate the following weather variables experienced by each trap during its active period: average temperature, average humidity (percent), average wind speed (mph), average barometric pressure, and accumulated precipitation. We then used generalized linear models in R (*lme4 package*), first creating univariate models for all weather models against the diversity value to determine significance, and then adding the habitat covariates. Site ID was included as random effect, with year and region included as a constant fixed effect, and all other variables being treated as fixed effects. Models were considered significant if their AIC_c score was >2.0 AIC_c less than the top candidate model and their variance inflation factors (VIF) were less than 4. Significant variables were combined through backwards stepwise selection to determine optimum variable set, and the top candidate model was chosen with the lowest AIC_c, VIFs, and highest adjusted R² value (Borcard et al. 2011).

Additionally, to determine how structural vegetation affected specific groups within the pollinator community, we will be employing a constrained ordination redundancy analysis (RDA) with a Hellinger transformation to account for community variation (Legendre & Gallagher 2001). RDAs will be conducted on the morphospecies data from the visual surveys

and are not included in this annual report but will be available in an upcoming peer-reviewed publication (Mathis et al. *in prep*).

Floral Community Analysis

We investigated associations between the floral community metrics (“used” abundance and diversity) and the structural vegetation and patch covariates to determine if certain habitat characteristics supported abundant and diverse floral communities. We created basic univariate linear models in program R with the function *lm*. This was done independently for each round to look for any dynamic associations between the floral community selected by pollinators and habitat characteristics across the summer. Models were considered significant and competing if their AICc score was >2.0 AICc greater than the null model.

Results

Structural Vegetation within Sites

Overall, most of our sites contained moderate forb cover and large sapling cover (**Fig. 2.5**). 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 more small sapling cover, bare ground cover, and grass cover, given that there is higher light availability for the understory herbaceous layer and the saplings have not had adequate time since harvest to grow larger than one meter (**Fig. 2.6**). Older sites had more large sapling cover, large shrub cover, and fern cover.

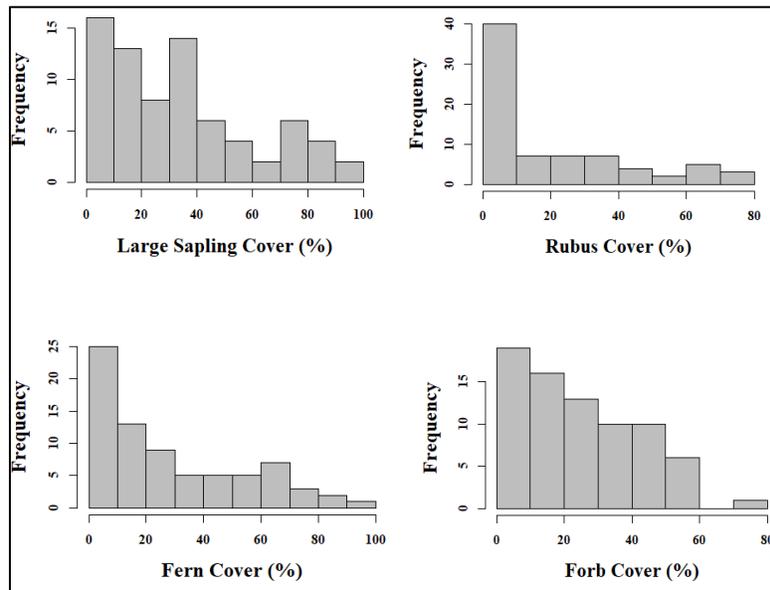


Figure 2.5. Histograms of vegetation features measured on the 100m vegetation transects. Values are expressed as the site-wide average of percent cover for each of the survey locations across the Central Appalachian Region of Pennsylvania.

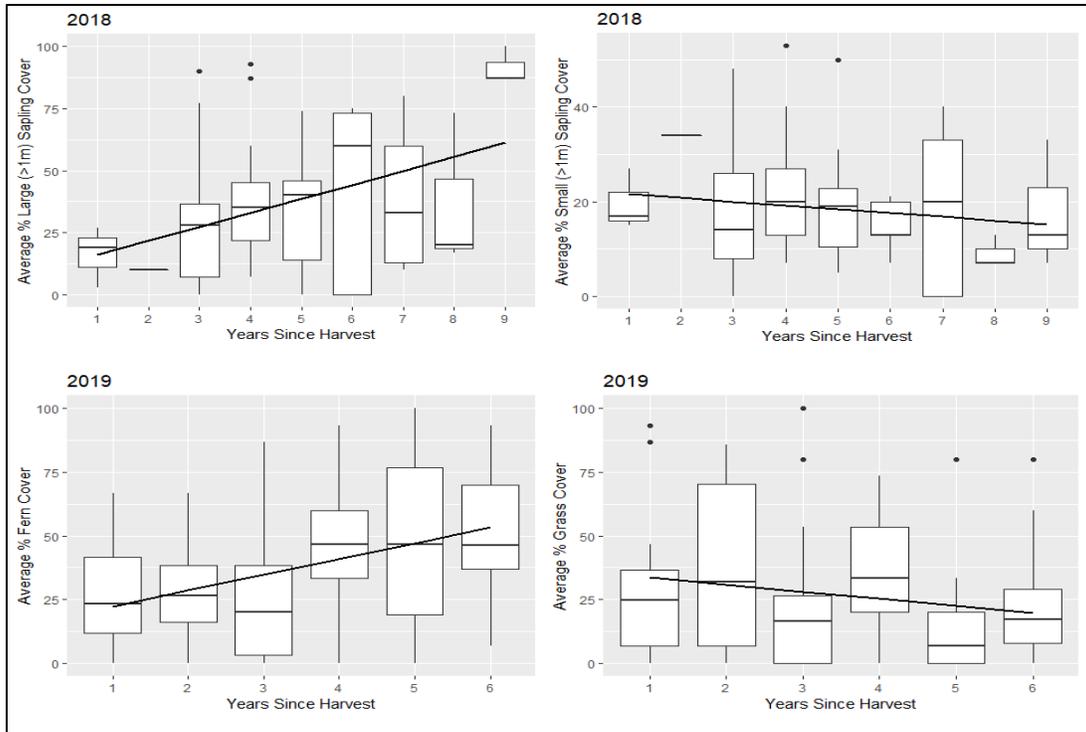


Figure 2.6. Boxplots of structural vegetation measured on our sites as a function of stand age (years since harvest), with some 2018 covariates above and some 2019 covariates below. These associations were consistent across years. Values are expressed as the site-wide average of percent cover for each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

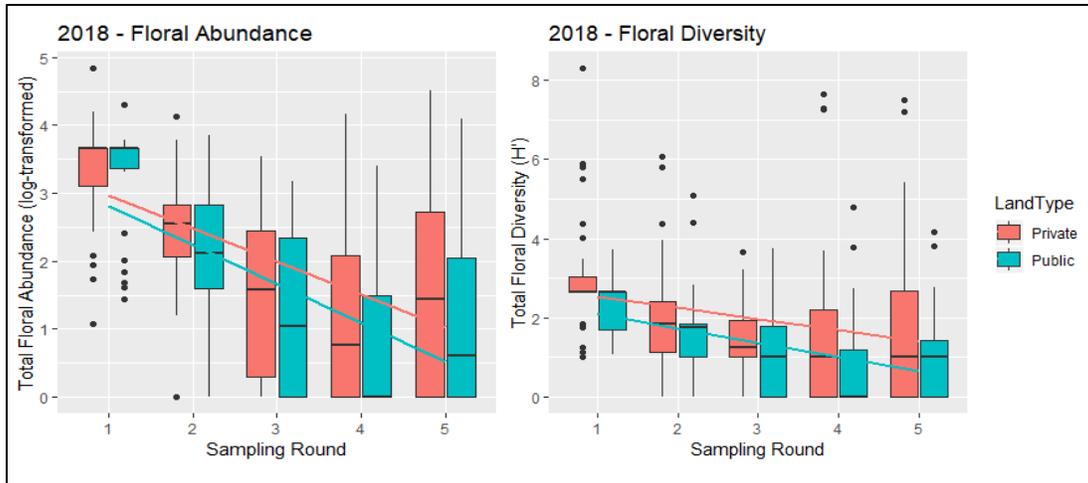


Figure 2.7. Floral Resources from 2018, grouped by landowner type. Surveys were every 3 weeks from May 14 – August 22, 2018. Floral abundance (left) is expressed as a log-transformation of total floral counts for each site. Floral diversity (right) is expressed as the floral diversity index (H') for 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 (≤ 6 years post-harvest) which tend to have more abundant and diverse floral communities.

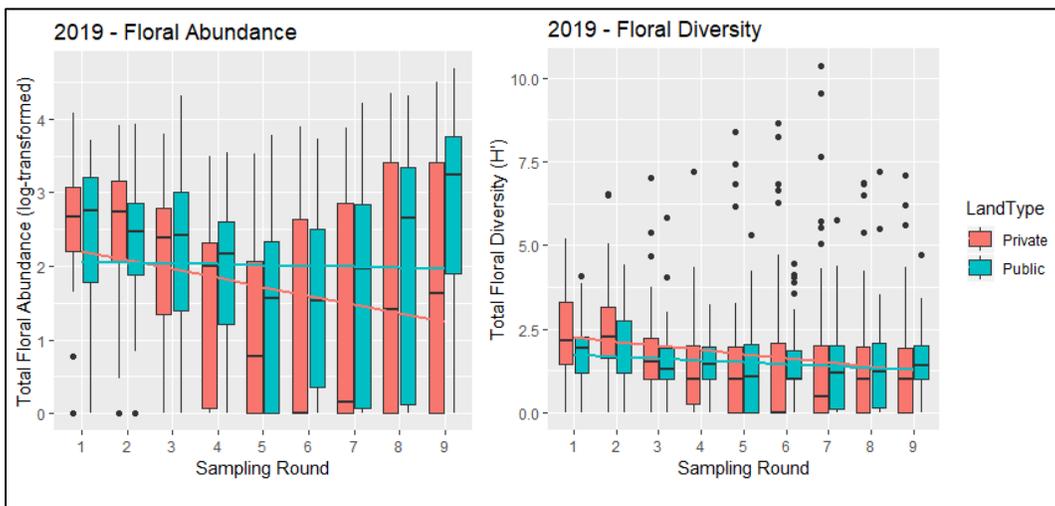


Figure 2.8. Floral Resources from 2019, grouped by landowner type. Surveys were every 2 weeks from May 14 – September 13, 2019. Floral abundance (left) is expressed as a log-transformation of total floral counts for each site. Floral diversity (right) is expressed as the floral diversity index (H') for 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	
Round 1 (May 14 – May 30)		May 14 – May 30	
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>
Round 2 (May 31 – June 22)		May 31 – June 22	
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>
Round 3 (June 23 - July 13)		June 23 – July 13	
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>
Round 4 (July 14 - Aug 1)		July 14 – Aug 1	
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>
Round 5 (Aug 2 - Aug 22)		Aug 2 – Aug 22	
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>
No Survey (Aug 23 – Sept 13)		Aug 23 – Sept 13	
-	-	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).

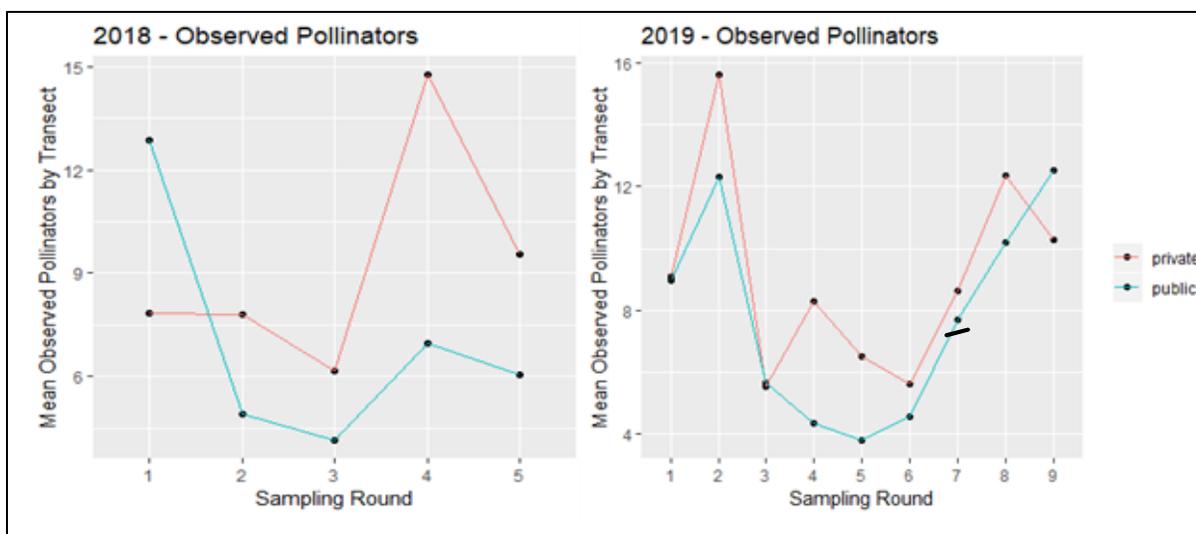


Figure 2.9. Mean pollinators observed per transect across the summer, grouped by land ownership. In 2018 (left), we conducted visual pollinator surveys every 3 weeks for a total of 5 sampling rounds (May 14 – Aug 22nd). In 2019 (right), we conducted visual pollinator surveys every 2 weeks for a total of 9 sampling rounds (May 14 – September 13th).

In 2019, we conducted visual transect surveys at 100 sites every two weeks for a total of nine visits each (weather permitting). Over these surveys, we observed >6,900 pollinators. Of these, the majority (n=5,728) were bees and 1,214 were butterflies. The most abundant morphospecies observed was bumble bees (n=2,494; 35.9%), followed by the small black bees (n=1,312; 18.8%) and butterflies (n=1,214; 17.5%). Most pollinator observation occurred in May (sampling round 2; **Fig. 2.9**), with another large observation event occurring in late August (sampling round 8). We frequently observed more pollinators on privately-owned sites than on publicly owned sites. On our sites where we conducted pollinator collections (public: n=20; private: n=20), we collected almost 1,300 bees and 146 butterflies across the entire sampling period. The bees collected represented 27 genera and 109 species, with the most common genera being *Dialictus* (n=276), *Bombus* (n=264), and *Ceratina* (n=232).

Pollinator Abundance 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. 2.10; Table 2.2 to 2.4**). This is likely because many butterflies feed on a variety non-floral of foods including fruit and animal scat, as well as being more mobile across an entire landscape. Butterflies may also be more closely tied to the availability of caterpillar host plants, a variable not considered in our analyses. Many bees, on the other hand, rely heavily on pollen/nectar as food for both themselves and their developing young (Michener 2008), and use these foraging resources often within 1 km of their nesting sites (Zurburchen et al. 2010). 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. Another important component of a healthy floral community is diversity. Our models suggested that average standardized diversity (H') was a significant predictor of density for both bees and butterflies. 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.

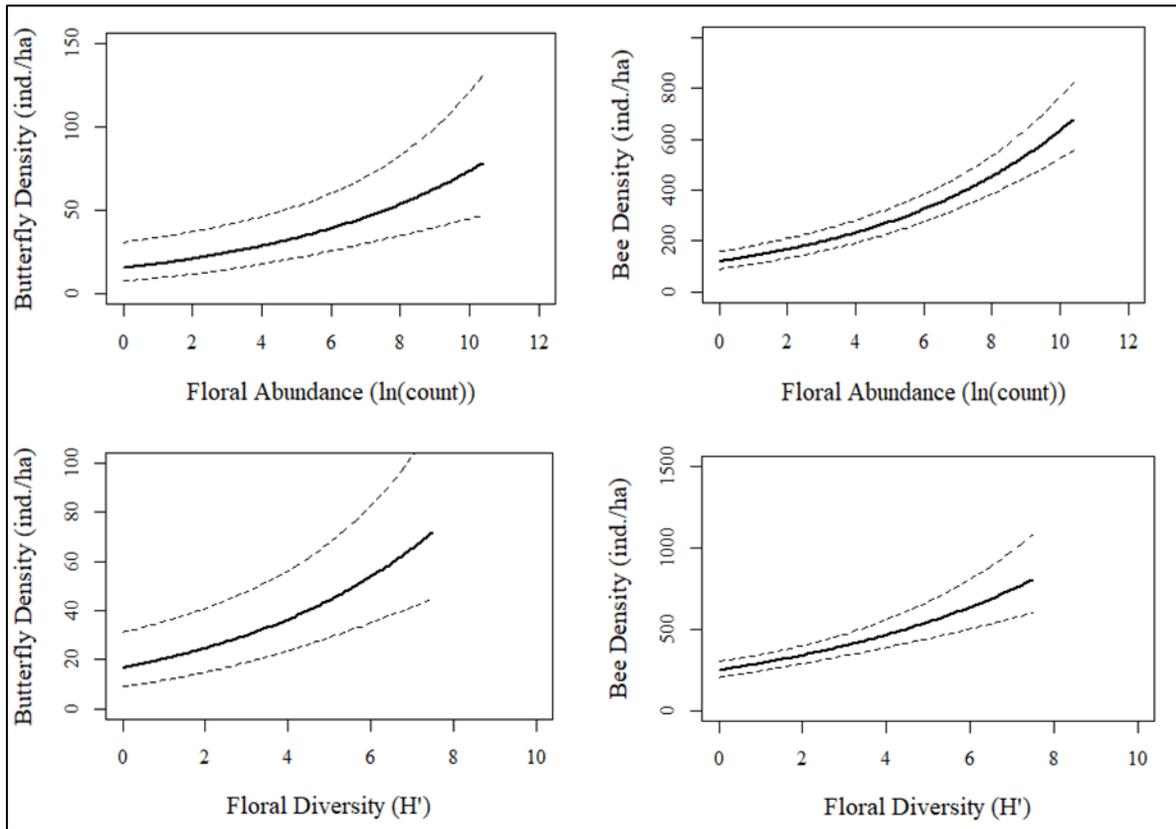


Figure 2.10. Estimated pollinator densities (individuals per hectare; butterflies on the left, bees on the right) by the average flowers on a site (log-transformed; above) or average floral diversity (standardized Shannon-Wiener Index H'). The dark line shows our model predictions and the dashed lines depict 95% confidence intervals. The shown models are fit to data from the fifth round of sampling in 2018 (Aug 2-22, 2018).

Table 2.2. 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.

2018 Pollinator Habitat Associations										
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)					-		-		-	-
<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 2.3. 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')	+	+			+	+	+	+	+

Using our visual pollinator survey data, we also created floral resource variables that included only the floral resources that pollinators selected for (*i.e.*, we observed pollinators nectaring on them). When we construct models with these variables, we get more significant results with narrower confidence intervals (**Fig. 2.11**). This suggests that pollinator survey methodologies in the future should include an observational survey as well as passive sampling to best determine associations between the floral community, structural vegetation, and pollinator communities.

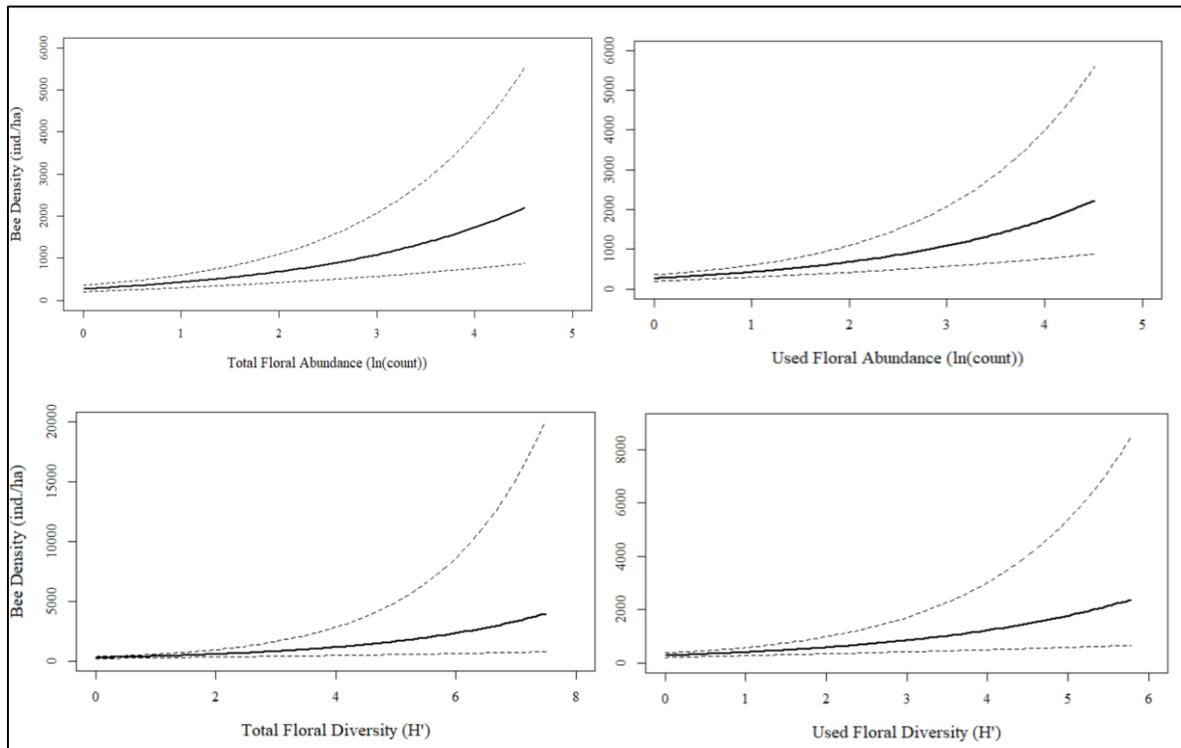


Figure 2.11. Estimated bee density (individuals per hectare) by floral abundance (log-transformed; above) and floral diversity (standardized Shannon-Wiener Index H'). Left graphs were created using variables with all floral resources and the right graphs were created using variables with only floral resources selected for by pollinators. 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 in 2018 (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 2.2**).

During many sampling visits, canopy cover, large (>1m) sapling cover, and large (>1m) shrub cover were negatively associated with pollinator density (**Table 2.2 to 2.4, Fig. 2.12**). 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 2.1**). As a result, small (<1m) shrubs were positively associated with bee density during the visit 1 analysis (estimate= 0.08 ± 0.04 , $p=0.03$), although this model competed with the null model. However, once those shrubs stopped flowering (visit 2+), density

was negatively associated with small 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 non-significant association with pollinators (estimate=-0.06±0.04, $p=0.15$). However, once *Rubus* spp. began to flower, we observed a positive association, primarily when *Rubus* was the only floral resource available to pollinators. Forb cover was an important predictor of pollinator abundance in many sampling rounds. This makes sense because the ‘forb’ category consisted almost entirely of wildflowers that produced floral resources during our sampling (e.g., *Solidago* spp., *Euthamia* spp., *Oxalis* spp., etc.). However, in our 2019 field season, observed that bees were negatively associated with forb cover in our fourth sampling round (**Table 2.4**). We expect that this occurred because our “forb” structural vegetation category was created including forbs that may not be flowering, and our fourth sampling round occurs during a well-known time during the summer where there is a dearth of floral resources. There is often a lot of forb cover, but it is comprised of spring flowers that are no longer flowering and fall flowers that have not yet begun to flower. We observed during that same visit, floral abundance was still positively associated with pollinator density (**Table 2.3 & 2.4**), so the pollinators were using what is available to them. We also noted that coarse woody debris and leaf litter were never significant predictors of pollinator abundance. This is likely because nesting materials (i.e., coarse woody debris for wood-nesting species, bare ground for ground-nesting species) are not limiting in most regenerating forests. We expected that the limiting resource, and therefore the resource that would be driving habitat selection within this landscape, would be foraging resources.

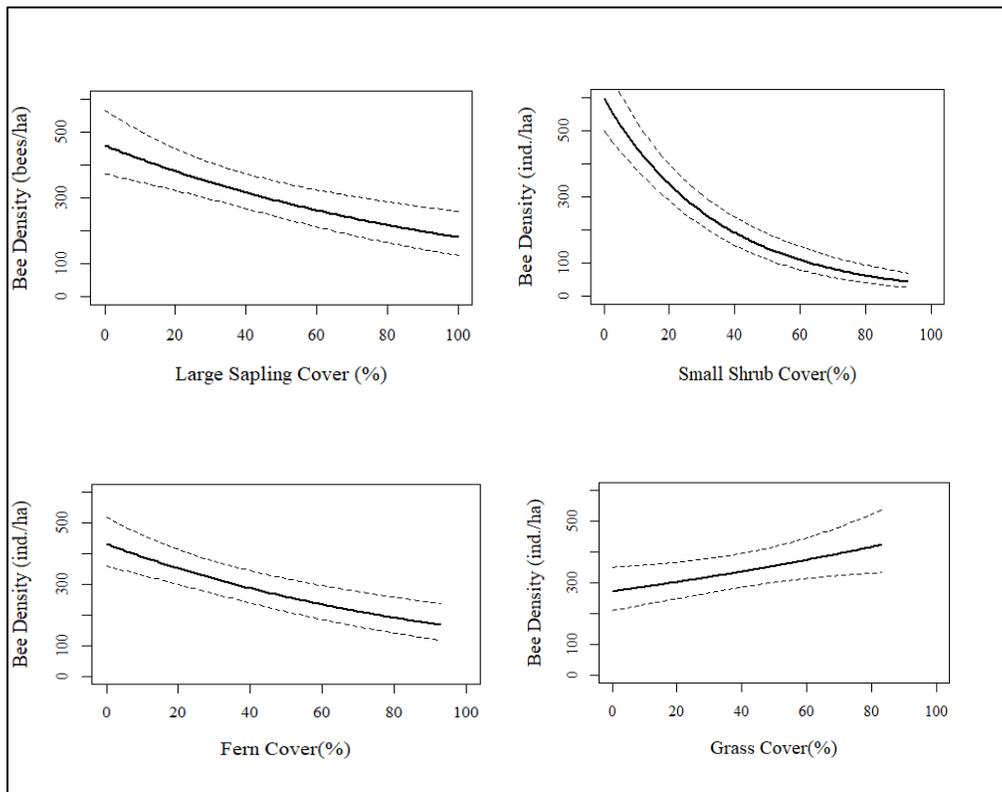


Figure 2.12. Vegetation models of bee density as a function of habitat features. All graphs shown include models with slopes discernable from zero. The dark line shows our model predictions and the dashed

lines depict 95% confidence intervals. The shown models are fit to data from the fifth round of sampling in 2018 (Aug 2-22, 2018).

For both bees and butterflies, site age (number of growing seasons, post-harvest) was a strong negative predictor of density later in the summer, with oldest sites having the lowest densities (**Fig. 13**). For example, in the fifth sampling round of 2018, 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.

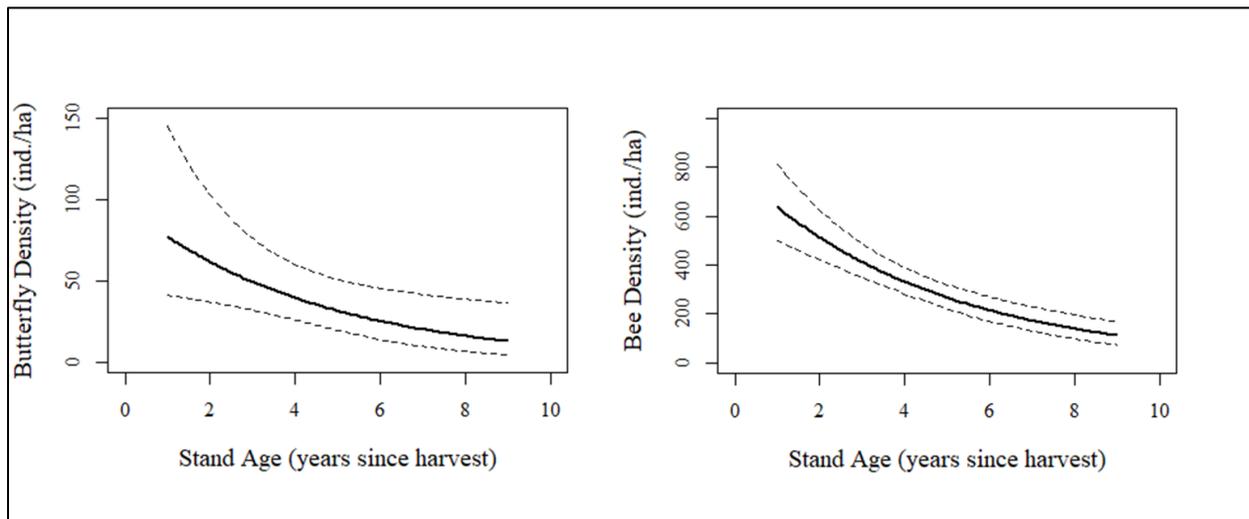


Figure 2.13. Estimated pollinator densities (individuals per hectare; butterflies on the left, bees on the right) as a function of the timber stand age (# of growing seasons). The dark line shows our model predictions and the dashed lines depict 95% confidence intervals. The shown models are fit to data from the fifth round of sampling (Aug 2 – 22, 2018).

Pollinator Diversity Modeling Results

Previous studies showed that species richness of flowering plants was the best predictor for species richness of bees in grasslands (Steffan-Dewenter & Tscharntke 2000; Taki et al. 2013). Given this, we predicted that floral diversity, as well as structural vegetative communities that promote high floral diversity, would be the best predictors of pollinator diversity within young forests.

To investigate what impacts pollinator diversity, we constructed linear mixed-effects regression models for our weather variables, patch characteristics, floral community metrics, and structural vegetation. Average wind speed (mph) and the ordinal date of the trapping event were the most significant predictors of pollinator density, both with negative associations (**Fig. 2.14**). This means that on windier days our collection efforts would result in fewer and less diverse collections. Additionally, pollinator diversity decreased as time went on through the summer, which makes sense given the known emergence times of pollinators. Considering the patch characteristics, our top model only included standage, which was a negative predictor of pollinator diversity (**Fig. 2.14**). As expected, floral diversity was a significant positive predictor

of pollinator diversity. Additionally, the structural vegetation covariates that were most important to pollinator diversity were large (>1m) sapling cover (negative association) and forb cover (positive association). This makes sense because large saplings are consistently negatively associated with pollinators (**Table 2.2 to 2.4, Fig. 2.12**), and if there are fewer pollinators you would expect a less diverse pollinator community, as well. When placed into a multivariate model, the variables included in the top model were average windspeed (mph), ordinal date, large (>1m) sapling cover, and forb cover.

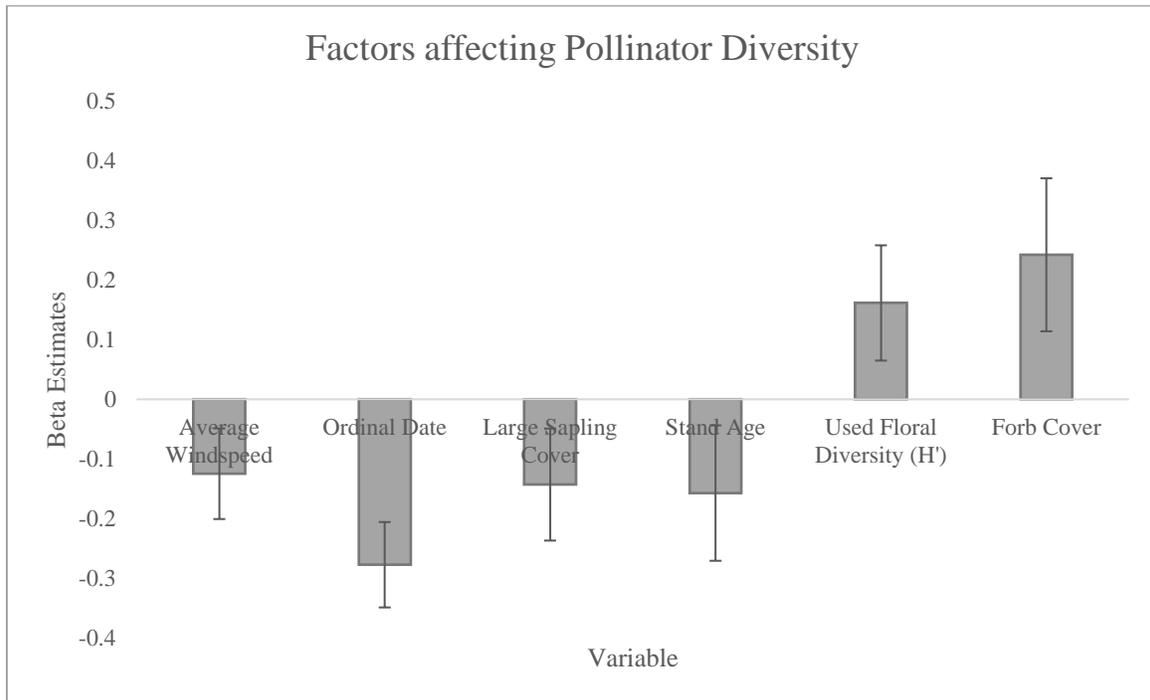


Figure 2.14. The most important variables influencing pollinator diversity, as determined by our linear-mixed effects regression models. Beta estimates are taken from univariate models for stand age and floral diversity, and from the multivariate top model for all remaining variables. Error bars are 95% confidence intervals ($1.96 \times \text{standard error from models}$).

Floral Community Modeling Results

Since we determined that foraging resources are likely most limiting in this environment, we investigated what structural vegetation and patch characteristics impacted the floral community in our sites. Since we also determined that the “used” variable category was a more significant predictor for pollinator communities (**Fig. 2.11**), we ran our models on that variable set.

We first investigated static associations (those which do not change throughout the growing season) between the mean used floral abundance and structural vegetation. We found that floral abundance was always positively associated with grass cover, forb cover, *Rubus* cover, and bare ground, with grass cover being the most significant predictor (**Fig. 2.15**). We also found that floral abundance was always negatively associated with leaf litter cover and canopy cover.

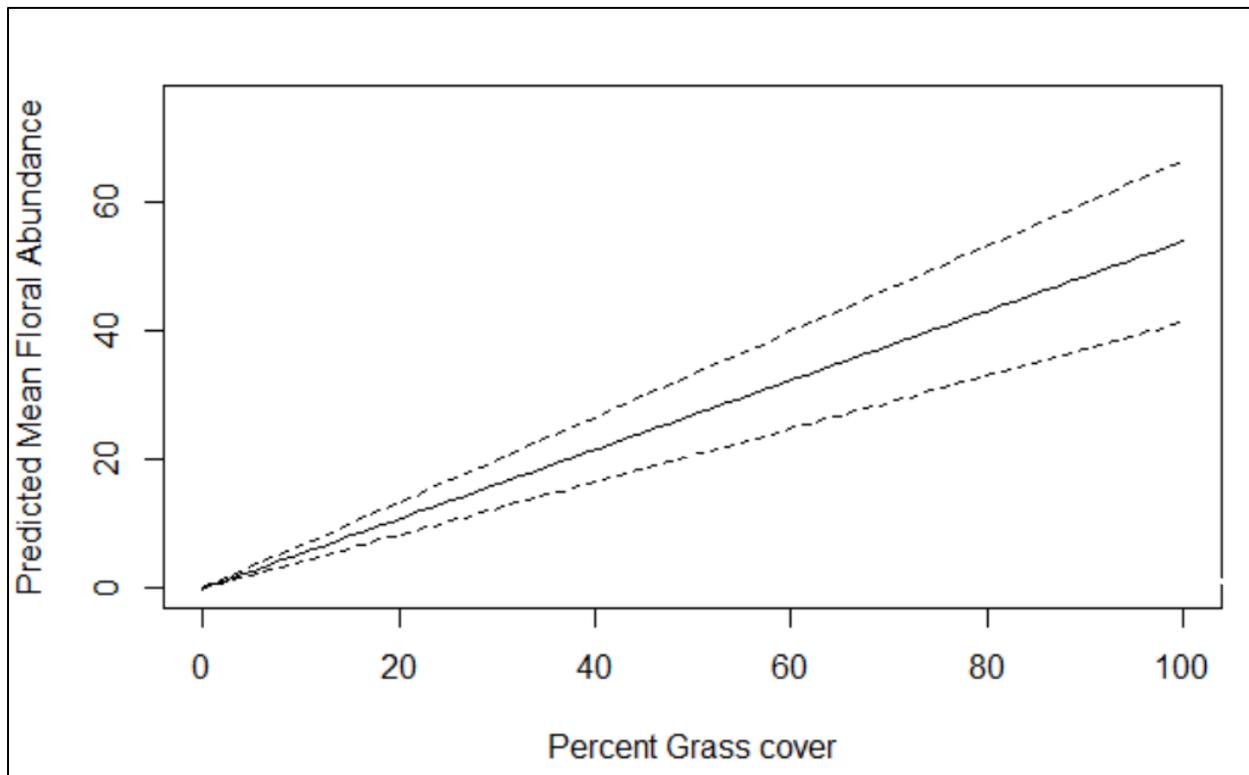


Figure 2.15. Results of our linear model with grass cover as a positive predictor of mean floral abundance. The dark line shows our model predictions and the dashed lines depict 95% confidence intervals. This model was created with a unique mean floral abundance variable for each site across the entire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

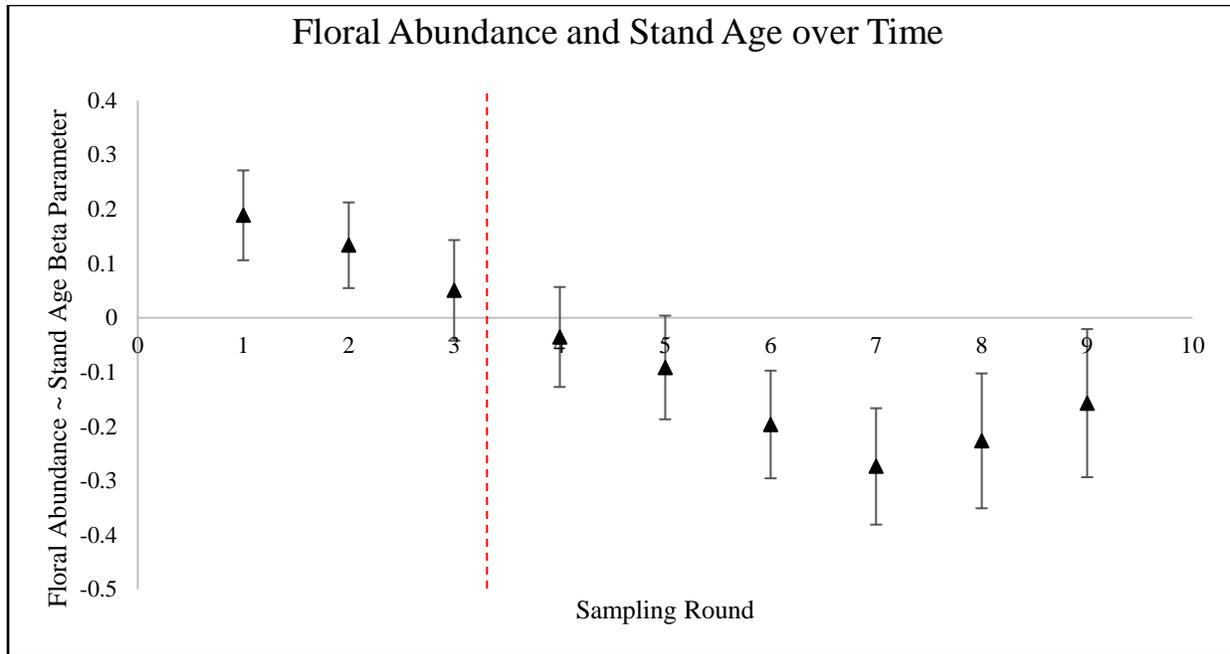


Figure 2.16. Beta parameter estimates from our lmer model of floral abundance ~ stand age, with associated standard error that these models generated. Sampling Rounds in 2019 are as follows: Visit 1: May 14 – May 27; V2: May 27 – June 9; V3: June 9 – June 24; V4: June 24 – July 8; V5: July 8 – July 24; V6: July 24 – Aug 4; V7: Aug 4 - Aug 17; V8: Aug 19 – Aug 31; V9: Aug 31 – Sept 12. Also added is a red dotted-line representing when most trees were expected to have full leaf-out (mid-June: based on the National Phenology Network).

Management Implications

Our results suggest that early-successional habitat created via overstory removal provides habitat for a dense population of pollinators for up to six 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 site age on pollinator densities. As expected, all pollinators benefit from a more diverse floral community. For bees, abundant floral resources were clearly of enormous importance, but this pattern was less clear for butterflies. Younger sites had more diverse pollinator communities, with pollinator diversity increasing with floral diversity and forb cover and decreasing with stand age and large sapling cover. More open sites (*i.e.*, more grass cover, bare ground cover, less canopy cover) had more abundant floral communities throughout the entire growing season. In the spring, older sites have more abundant floral resources because the spring floral communities have had more time to establish post-disturbance when compared to younger sites. In the summer after leaf-out, these same sites have relatively little floral resources compared to the younger sites that have higher light availability and more open space for floral resources.

Our results suggest that ensuring an abundance of early-successional habitat (< 6 years post-harvest) within heavily forested landscapes will likely benefit native pollinators within those landscapes by providing abundant floral and nesting resources across the entire growing season. This could be accomplished by maintaining dynamic forest landscapes where harvests are rotated through time and forest age classes are diversified – leaving some early-successional habitat for pollinators on the landscape at all times. Programs like the WLFW Golden-winged

Warbler partnership are creating young forests that provide valuable foraging resources for pollinators where these resources would otherwise not exist without management. Additionally, sites created by *WLFW* and managed for the Golden-winged Warbler often had more abundant and diverse floral communities than nearby sites on State Game Lands and State Forests, proving that this management is valuable to creating habitat on the landscape that is beneficial to abundant and diverse pollinator communities.

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Part III. A comparison of American Woodcock use of early successional communities associated with timber harvests and old field management and development of a distance-based protocol using thermal imagery to estimate nest density.

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Background

The loss and degradation of early successional communities in eastern North America is well-documented (Thompson et al. 2013; King & Schlossberg, 2014). This reduction in the availability of early successional communities has been attributed to several factors including progression of natural succession on abandoned farmland through the 19th century, increased urbanization and industrialization, changing forest management goals and methods, and disturbance suppression (Hunter et al. 2001). Due to anthropogenic interference in natural drivers of ecological succession, it is unlikely that enough early successional communities will be available to sustain viable populations of many disturbance-dependent species (Askins 2001; King & Schlossberg 2014). The implementation of habitat management actions by government agencies, conservation organizations, and the forest industry that create and maintain early successional communities on public and private lands in eastern North America will be critical for stabilizing declining populations of many disturbance-associated species (DeGraaf & Yamasaki 2003).

In the Northeastern U.S., the loss of early successional communities (-2.4% annually; King & Schlossberg 2014) is thought to be the predominant factor driving population declines of species associated with early successional habitats like the American Woodcock (*Scolopax minor*; -1.1% annually) (Palmer 2008; Seamans & Rau 2018). Long-term monitoring through the United States Fish and Wildlife Service (USFWS) reveals that American Woodcock populations in the eastern management region are declining by 1.18% annually (Seaman & Rau 2018). This decline has been, in part, attributed to the reduced availability of breeding habitat (Dwyer et al. 1983). Indeed, many of the species' life history phases such as courting, brood rearing, and nesting are dependent, in part, on the availability of early successional communities (Capel et al. 2008; Bakermans et al. 2015). The long-term population declines of woodcock and its popularity as a game species has resulted in several efforts to implement habitat management guidelines to benefit this species (Kelley et al. 2008; Capel et al. 2008; Bakermans et al. 2015). Monitoring efforts that evaluate woodcock response to habitat management on public and private lands will help quantify the extent to which these efforts contribute to the species' recovery. Additionally, such work also allows for comparisons of woodcock use of early successional communities originating from different management contexts (*i.e.*, old field management vs. timber harvesting). Ultimately, insight from monitoring is critical for employing an adaptive management approach for conservation efforts intended to benefit this species.

Across their range, woodcock breeding populations are primarily monitored annually through a long-term program coordinated by the U.S. Fish and Wildlife Service in cooperation with state agencies (Seamans & Rau 2018). The USFWS Singing Ground Survey (SGS) was developed to generate annual population estimates for woodcock using a standardized protocol based on the conspicuous courtship display of males during the spring breeding season (Duke 1966). The USFWS SGS is a road-based survey and thus not designed to evaluate woodcock response to habitat management at local scales (*i.e.*, within a managed patch; Tavernia et al. 2018). Some monitoring efforts have used modified versions of the SGS protocol to evaluate woodcock response to local habitat management (*i.e.*, Bakermans et al. 2015). Such surveys can provide valuable data that help public and private land managers make more informed management decisions and generate public support for conservation efforts that target the species (Capel et al. 2008; Tavernia 2018; Weber & Cooper 2019). Another limitation of the SGS is that it only evaluates singing male abundance and lacks the ability to quantify other important demographic parameters such as female density and reproductive success. Such information is important for gaining a more comprehensive evaluation of local population response to habitat management. Moreover, evaluating which habitat management practices produce microhabitat and local landscape conditions that are optimal for woodcock productivity can be used to improve existing guidelines and more effectively use limited conservation funding.

Beyond counting singing males, research and monitoring of American Woodcock is difficult due to the species' cryptic behavior and affinity for dense woody cover. Previous studies have examined woodcock breeding ecology using methods that involve nest searching, capturing and tagging adults, and using pointing dogs to capture and band fledglings (McAuley et al. 1993; Shuler et al. 1986; Wiley & Causey 1987). Such approaches are time consuming, invasive, and do not account for detection error. The development of sampling protocols that employ thermal technologies may improve the effectiveness and efficiency of studies examining American Woodcock breeding ecology. Thermal technologies are growing in their use as tools to study and manage wildlife (Ditchkoff et al. 2005; Blackwell et al. 2006; Gauthreaux & Livingston 2006; Christiansen et al. 2014). In fact, hand-held thermal technology was recently used to examine the timing of American Woodcock nest initiation (Keller 2016). Attempts to develop survey protocols that utilize hand-held thermal technology in a distance sampling framework that accounts for imperfect detection is warranted. Collectively the use of a modified singing ground survey protocol and a thermal-based distance sampling protocol can provide considerable insight to the use and relative quality of managed habitats by male and female American Woodcock.

Since 2012, government agencies, conservation organizations, universities, and private landowners have partnered to manage thousands of acres of forest and shrublands on public and private lands in Pennsylvania to increase habitat availability for disturbance-dependent species such as American Woodcock, Ruffed Grouse, and Golden-winged Warbler (*i.e.*, McNeil et al. 2016; Lutter et al. 2019; McNeil et al. *in press*). The management practices used in Pennsylvania to create or maintain early successional communities can be placed into two general categories: silvicultural (*i.e.*, regenerating timber harvests) and agricultural-derived (*i.e.*, managed old fields/shrublands). Vegetation structure, plant species composition, and soil conditions of early successional communities that develop in association with forest regeneration and old field

succession differ considerably (Vitousek et al. 1989; DeGraaf & Yamasaki 2003; Tatoni & Roche 1994). As such, it is reasonable to postulate that old fields and regenerating timber harvests differ in their quality as breeding habitat for American Woodcock. We initiated a monitoring effort to evaluate woodcock use of silviculture-derived and agriculture-derived early successional communities in Pennsylvania. Specifically, we used a) a modified SGS protocol to quantifying male density and b) developed and tested a distance-based thermal imagery protocol to quantify woodcock nesting density within managed early successional communities in Pennsylvania. Herein we report preliminary findings from the first year of this effort.

Objectives

The overall objective of this CEAP component is to compare American Woodcock use of early successional communities resulting from timber harvests and old field management in Pennsylvania. Specific project objectives include:

1. Compare occupancy and density of American Woodcock in silvicultural-derived and agricultural-derived early successional communities in Pennsylvania.
2. Examine the potential for using thermal imagery to quantify American Woodcock nesting density and nest site selection in managed early successional habitats.

Methods

Study Area

We monitored American Woodcock use of managed habitats in western Pennsylvania (**Fig. 3.1**). Managed areas we surveyed were primarily the result of two management practices; regenerating timber harvests and old field management. We surveyed singing woodcock across 59 unique patches of early successional communities (36 timber harvests and 23 old fields; **Table 3.1**) that were managed between 2010 to 2019. Regenerating timber harvests were the result of commercial overstory removal harvests and ranged in size from 4-161 ha. Old field management sites were created using forestry mulchers to remove high densities of small woody stems, tree cutting, and brush hogging, or allowed to undergo natural succession post-agricultural use. Old field sites ranged in size from 11-112 ha.

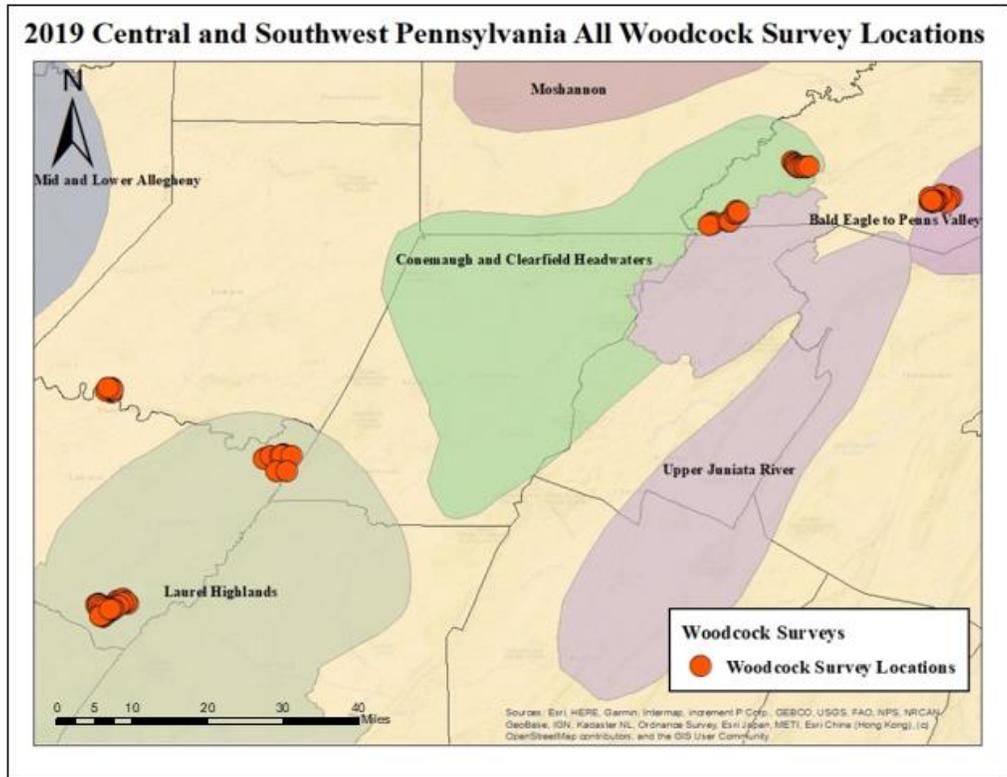


Figure 3.1. Locations of sites in western Pennsylvania where American Woodcock surveys were conducted during April – May 2019. Portions of these sites were managed in recent years to create or enhance early successional communities. Site locations are shown in relation to several focal geographies (Laurel 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.

Table 3.1. Site-specific information for early successional communities in western Pennsylvania where American Woodcock surveys were conducted. Surveys occurred between 15 April – 15 May 2019.

Site Name	Year Treated	Treatment Type	Number of Unique Patches	Total Area (ha)	Number of Growing Seasons	Number of Points
Forbes	2015	Old Field Management	2	12	4	3
Forbes	2016	Old Field Management	1	7	3	1
Forbes	2017	Old Field Management	4	21	2	6
Forbes	2018	Old Field Management	3	29	0	6
Indian Creek Farms	fallowed in 1997	Old Field Management	2	7	NA	2
Indian Creek Farms	2015	Old Field Management	1	8	4	2
Indian Creek Farms	fallowed in 1997	Old Field Management	2	7	NA	3
SGL 411	2018	Old Field Management	6	112	1	8
SGL 42	2017	Old Field Management	2	11	2	2
Old Field Management Totals:			23	214	NA	33
Skyview	2018	Overstory Removal	3	84	0	8
SGL 176	2019	Overstory Removal	7	102	0	13
SGL 176	2010	Overstory Removal	15	30	8	9
SGL 60	2018	Overstory Removal	3	48	6	6
SGL 33	2013	Overstory Removal	1	161	6	9
SGL 42	2016	Overstory Removal	2	4	3	2
SGL 42	2014	Overstory Removal	2	16	5	4
SGL 42	2010	Overstory Removal	2	4	11	2
Indian Creek Farms	2016	Overstory Removal	1	8	3	2
Overstory Removal Totals:			36	457	NA	55

American Woodcock Singing Ground Surveys

We conducted American Woodcock surveys during the predetermined safe date period (April 15th -May 5th) established by the United States Fish and Wildlife Service (USFWS 2018).

We modified the standard USFWS- American Woodcock SGS protocol to monitor woodcock at a local scale (*i.e.*, within managed sites). Using ArcGIS, each survey point was randomly placed within the boundaries of each early successional patch. All survey locations were at least 50 m from the patch boundary and spaced ≥ 250 m apart to avoid double counting of individuals (*i.e.*, see **Fig 3.2**). Start times for woodcock surveys were based on two factors: time of sunset and sky condition. We determined when woodcock surveys began by employing the formula “official local sunset time + sky condition time = start time”. If there was a clear or moderate overcast ($\leq 3/4$ overcast), 22 minutes (sky condition time) was added to the sunset time to determine the starting time. If there was high overcast ($>3/4$ overcast) 15 minutes was added to the sunset time. Extensive cloud cover causes darkness to occur sooner, so woodcock will display earlier on cloudy nights than on clear nights. Each survey lasted for four minutes and all surveys were completed no later than 38 minutes after the start time for surveys each evening. We recorded the number of singing (peenting) males during each survey. Consistent with USFWS SGS protocol, we did not conduct surveys on evenings with high winds, heavy precipitation, or if the temperature was below (4° Celsius. Any noise disturbances during a survey (*e.g.* frog chorus, loud bird calls, traffic) was recorded as either a low, medium, or high disturbance.

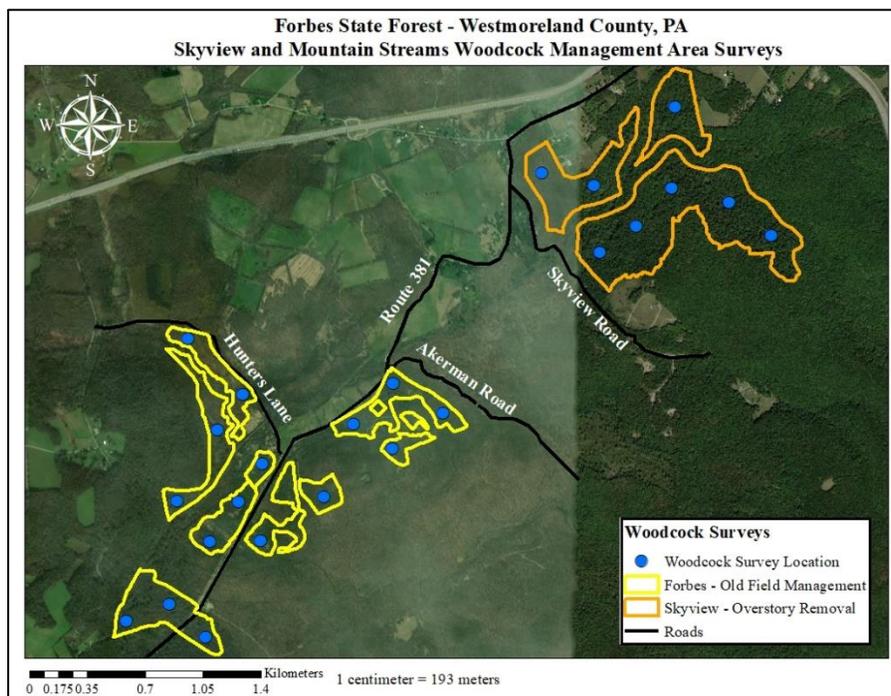


Figure 3.2. Example distribution of locations in managed early successional communities where American Woodcock singing ground surveys were conducted. The map shows 2019 survey locations (blue dots) within the Forbes State Forest Skyview and Mountain Streams Woodcock Management Area (MSWMA) in Westmoreland County, PA. Early successional communities were the result of old field management (yellow polygons) or timber harvests (red polygons). Survey locations were spaced >250 m from each other to avoid double counting of singing males.

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**).

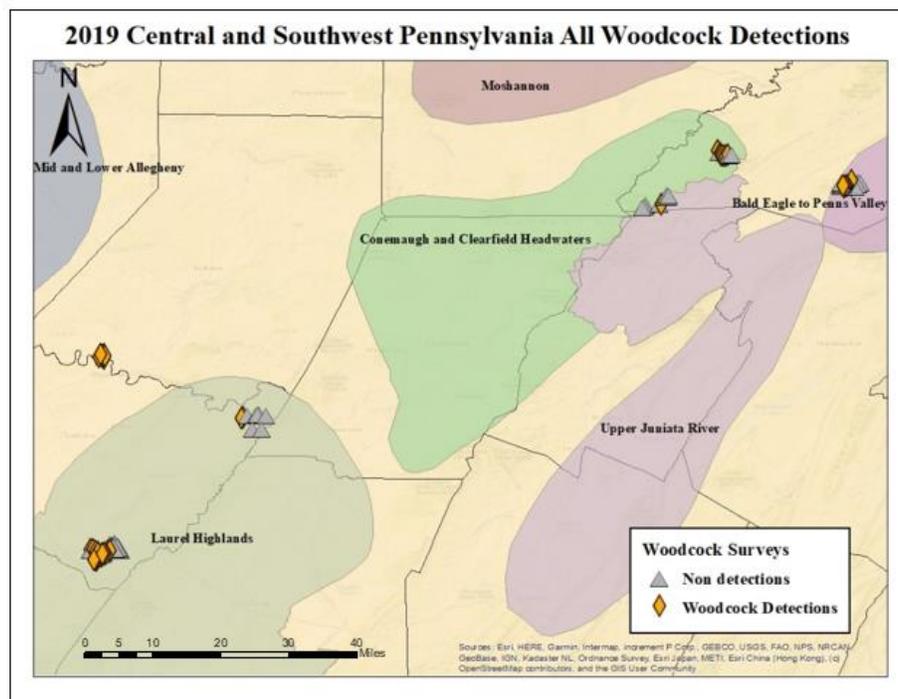


Figure 3.3. Survey locations in western Pennsylvania where American Woodcock were detected during singing ground surveys conducted between 15 April – 15 May 2019. These sites were managed in recent years to create or enhance early successional communities.

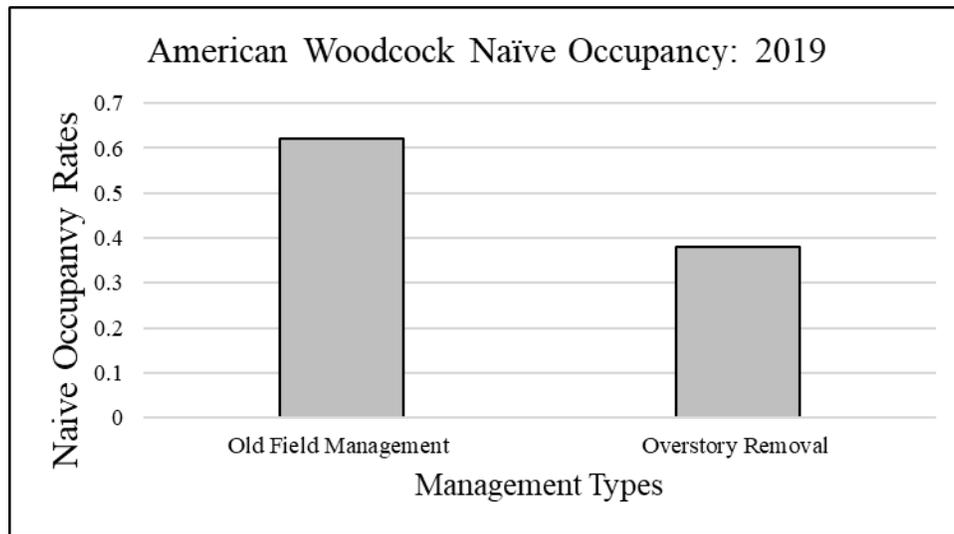


Figure 3.4. Naïve occupancy of American Woodcock at managed early successional communities in western Pennsylvania. Surveys were conducted between 15 April to 15 May 2019.

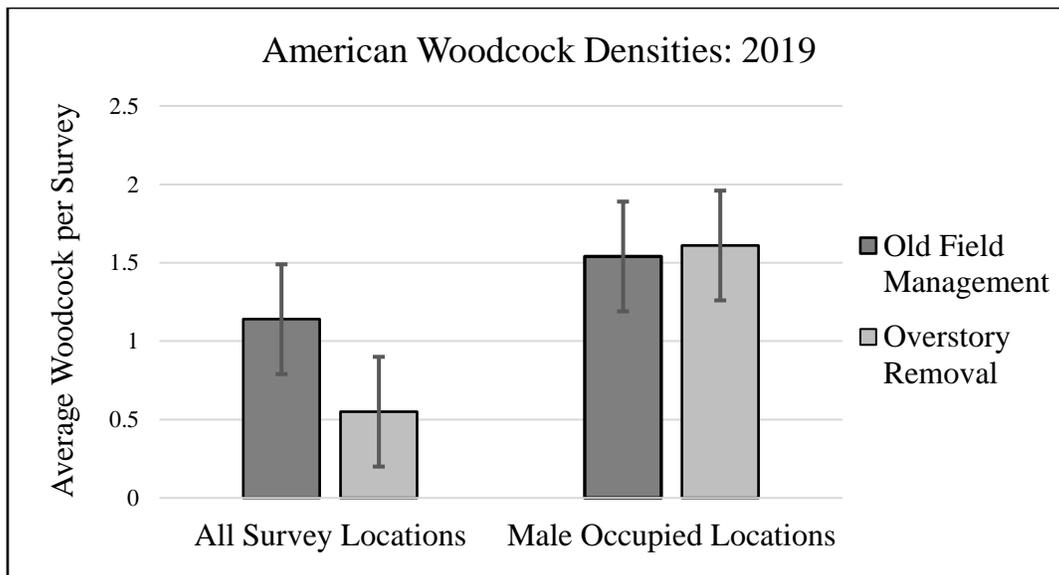


Figure 3.5. American Woodcock density for all survey locations and only survey locations where the species was detected. Surveys were conducted in early successional communities located in western Pennsylvania between 15 April – 5 May 2019.

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

detecting various species, but as conditions became darker, observer speed was reduced in the dense vegetation. While the effort we placed into the pilot thermal protocol was limited relative to other project activities, we were successful at using the thermal units to detect several wildlife species including American Woodcock, cottontails (*Sylvilagus* sp.), Ruffed Grouse (*Bonasa umbellus*), Eastern whip-poor-wills (*Antrostomus vociferus*), white-tailed deer (*Odocoileus virginianus*), North American porcupines (*Erethizon dorsatum*), Eastern Towhees (*Pipilo erythrophthalmus*), and Song Sparrows (*Melospiza melodia*), to name a few. With respect to woodcock, we detected two nesting females and six diurnal roosting individuals (not on nests); **Fig. 3.6**). Given our promising preliminary results, we have developed a distance-based survey protocol for hand-held thermal units. We are currently testing the protocol to generate detection probabilities using dummy woodcock and to train observers for protocol implementation which will occur from mid-March – early May 2020.

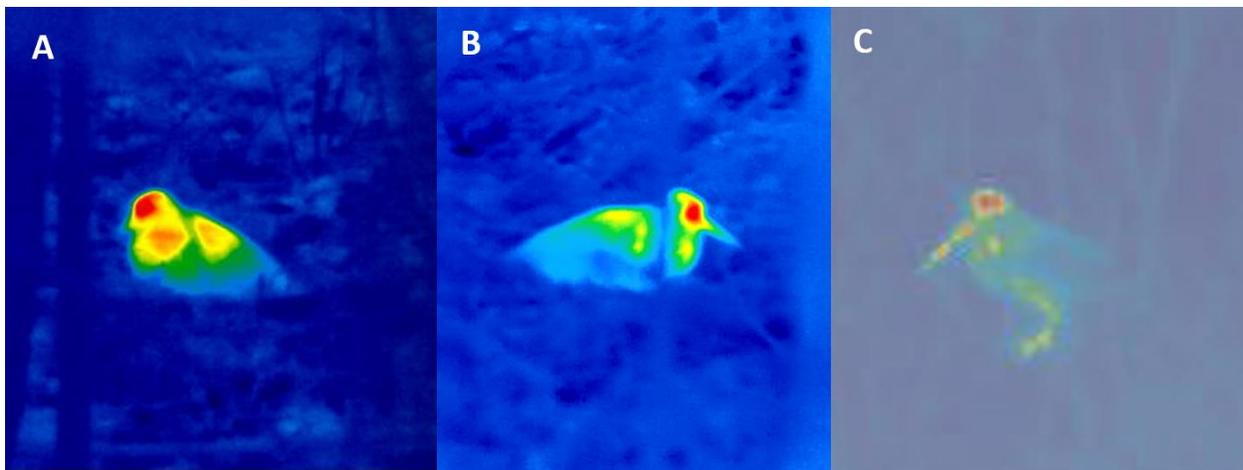


Figure 3.6. Thermal images of a) nesting woodcock in an old field managed sites; b) nesting female in regeneration timber harvest; and courting male in an old field managed site. Images were captured using a Pulsar hand-held thermal scope in April and early May 2019.

Other notable findings

The Eastern Whip-poor-will is a forest bird that also requires a mosaic of forest ages on its breeding grounds. This nocturnal member of the nightjar Order is experiencing population declines throughout eastern North America (Wilson & Watts 2008; Hunt 2013; Tozer et al. 2014; Sauer et al. 2014). Unlike woodcock, which have peak courting occurring during dusk and dawn, Whip-poor-wills begin singing at dusk with peak display occurring throughout the night (Sauer et al. 2014). As such, the Whip-poor-will survey protocol does not align perfectly with that of the American Woodcock SGS (Akresh & King 2016). Nonetheless, we opportunistically recorded Whip-poor-will detections during woodcock surveys. We detected Whip-poor-will (n = 61 individuals) at 26 of 88 points (30%). Whip-poor-will detections occurred at 5 of 33 (15%) old field management sites, and 21 of 55 (38%) timber harvests. The average number of Whip-poor-will at occupied points was 2.34 males/pt (95% CI: 1.99-2.71), with more males detected per

point in timber harvests ($n = 2.57$ males/pt; 95%CI: 2.2-2.9), than in old field sites ($n = 1.4$ males/pt; 95%CI: 0.72-2.08).

Conclusions and Future Efforts

Efforts to create or enhance early successional communities on public and private lands is an ongoing priority for many state and federal agencies and their conservation partners. Our 2019 monitoring effort at several recently managed early successional communities in western Pennsylvania revealed that woodcock occupancy (singing males) varied between silviculture-derived and agriculture-derived habitats. However, singing male density for occupied sites did not differ between silviculture-derived and agriculture-derived habitats. Our efforts in 2019 also revealed that thermal imagery has strong potential as a tool to survey nesting female woodcock as well as additional species like Eastern Whip-poor-wills. In 2020, we intend increase the number of singing ground surveys we conduct in silviculture-derived and agriculture-derived early successional communities. Sites we select for monitoring will include those managed through NRCS's Working Land for Wildlife partnership, Voluntary Public Access-Habitat Improvement Program, and commercial and non-commercial habitat projects on nearby public lands. We will also implement a thermal-imagery protocol at several silviculture-derived and agriculture-derived early successional communities. Ultimately our efforts are intended to provide conservationists with a better understanding of the relative contributions of silvicultural-derived and agriculturally-derived early successional communities.

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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

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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)(Russel 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; Rossell 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; Russel 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

Methods

Study area

We used 10 pairs of fenced and unfenced regenerating overstory removal harvests in Centre County in northcentral Pennsylvania (**Table 4.1**; pictured right). All timber harvests were on State Game Lands (SGL) managed by the Pennsylvania Game Commission: SGL 033 (7,100 ha), SGL 060 (3,540 ha), and SGL 100 (8,400 ha). Immediately upon harvest, fenced portions of the timber harvests were established using 2-m tall woven-wire fencing specifically intended to exclude deer. Unfenced portions of the harvests were directly adjacent to the fenced portions and were completely accessible to deer. Thus, we utilized a randomized block design where each pair of control (unfenced) and treatment (fenced) harvests had the same local landscape characteristics. Each pair of fenced and unfenced sites were harvested at the same time by the same operators between 2009 – 2012. Residual species composition and residual basal area (m^2/ha) for fenced (2.1 ± 0.58 ; mean \pm SE) and unfenced (3.3 ± 0.66) harvests were similar. Fenced timber harvests ranged in size from 3.4 – 57.9 ha (18.8 ± 5.9 ha) and unfenced harvests ranged from 6.5 – 50 ha (mean = 20.2 ± 4.9 ha). Fencing was removed at two study sites (Aikley Hollow and Firetower North) in 2018, less than one year prior to our study. These fences were removed after foresters determined the woody regeneration had achieved heights beyond the reach of deer. Additionally, one fenced/unfenced pair (322 Salvage) was treated with a prescribed fire in spring 2016. SGL 100 supports both white-tailed deer and elk (*Cervus elaphus*) populations.

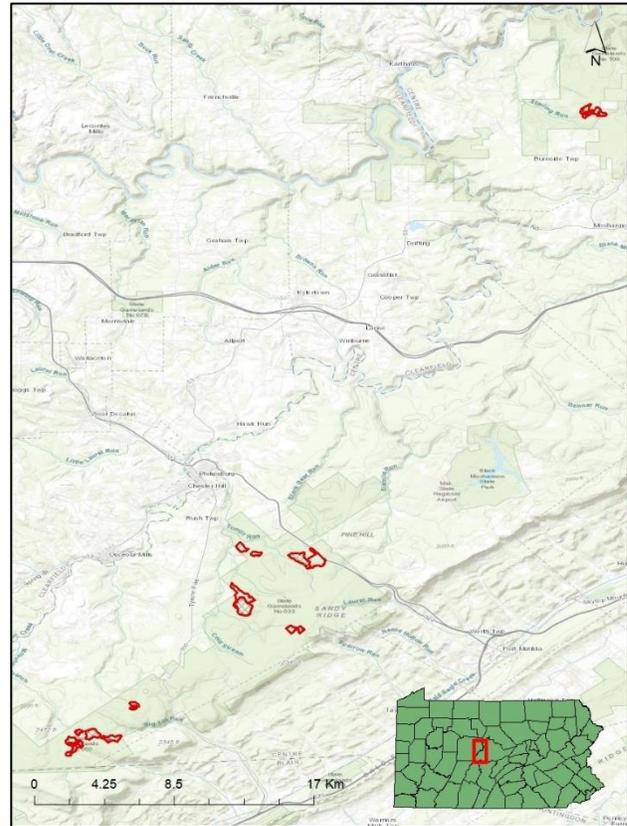


Table 4.1. Site descriptions for each of 10 paired fenced and unfenced overstory removal timber harvests in northcentral Pennsylvania. Vegetation and avian communities were surveyed in each site in 2019.

Note: RBA = residual basal area

SGL	Site Name	Size (ha)	Year Cut	Elevation (m)	RBA (m ² /ha)
100	Aikley Hollow Fenced ^a	13.3	2010	531	3.4
	Aikley Hollow Unfenced	24.6		530	4.2
60	Firetower N Unfenced	8	2009	689	7.7
	Firetower N Fenced ^a	4		700	0
	Firetower 1 Unfenced	15.9	2009	735	2.5
	Firetower 1 Fenced	3.4		740	0
	Firetower 2 Unfenced	6.5	2009	735	1.1
	Firetower 2 Fenced	8.1		732	3.4
	Firetower 3 Fenced	29.5	2009	713	0.3
	Firetower 3 Unfenced	23.1		678	3.4
	Firetower 4 Unfenced	6.7	2009	733	1.7
	Firetower 4 Fenced	4.7		762	2.3
33	Hawk Run Unfenced	44.5	2012	596	5.6
	Hawk Run Fenced	57.9		620	5.0
	Red Horse Unfenced	13.4	2009	688	1.7
	Red Horse Fenced	7.8		692	4.0
	322 Salvage Unfenced ^b	50	2012	649	2.3
	322 Salvage Fenced ^b	42.2		608	0.7
	Crows Nest Fenced	16.7	2012	576	1.7
	Crows Nest Unfenced	9.5		595	2.3

^a Exclosure fencing removed in 2018

^b Prescribed fire treatment in spring 2016

Browsing and vegetation surveys

We evaluated browse intensity along four parallel transects that ran the length of each timber harvest (**Fig. 4.1**). Transects and start/stop points for each transect were plotted in ArcGIS version 10.6. Transects were spaced equally apart in a manner that allowed us to evenly sample across each harvest. The initial transect starting point (*i.e.*, the first plot) was randomly selected using a random number generator to choose a number between 0 m and the maximum distance between points. Browse survey plots (1m radius circle) were evenly spaced along each transect and the number of plots varied for each harvest depending on harvest area. We established 20 plots for the first 4 ha and added one plot for each additional ha. For example, if a harvest was 17 ha we established 33 browse survey plots: 20 plots for the first 4 ha and 13 plots per additional 13 ha. Harvests that were less than 4 ha were sampled at 20 plots, similar to SILVAH sampling protocol (Brose et al. 2008). Plots were spaced equally along transects and the distance between

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

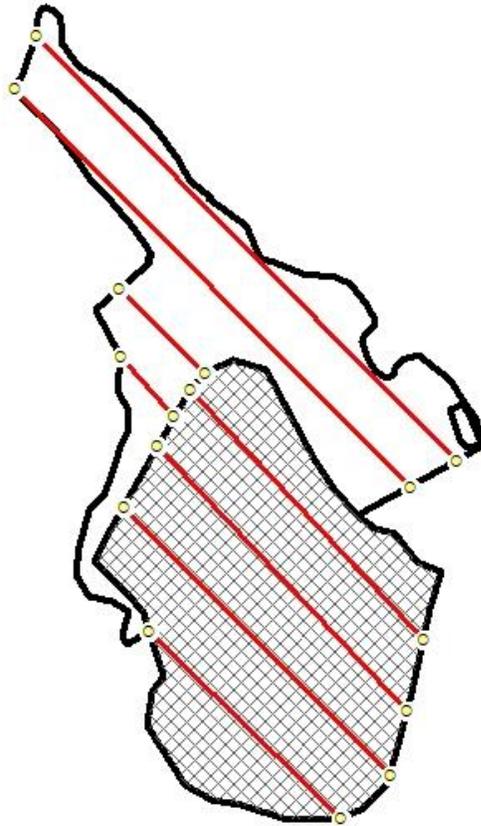


Figure 4.1. Layout of browsing transects (red lines) for one pair of fenced (hashed; 57ha) and unfenced (open polygon; 44ha) harvests on State Game Lands 033). Our 1m radius sample plots were evenly distributed along each transects and spacing was based on harvest size and transect length.

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

placed points in fenced and unfenced harvests, respectively. At each sampling point, we established three 5m² plots (5m x 1m). The starting location of each 5 m² plot was randomly located between 0 and 30m along 35 m long transects that radiated from point center at 0°, 120°, and 240° directions from each plot. Within each 5 m² plot we recorded the species name and height (m) of each woody stem. Horizontal and vertical vegetation density compared the mean proportion of squares >50% obscured by vegetation. We quantified horizontal vegetation using a 2m tall vegetation profile board that consisted of twenty 20 x 20 cm squares (Nudds 1977). The profile board was held by one technician 5 m from the plot center at the 0°, 120°, and 240° directions, while another technician viewed the density board from point center at a height of 1m and recorded the number of squares that were >50% obstructed by vegetation. This process was repeated at the end of a randomly chosen 35 m transect. We used a densiometer to quantify vertical vegetation density at four locations per point, once at point center and at the end of each of 35 m transect (Lemmon 1956).

We used an ocular tube (*i.e.*, GRS densitometer) to estimate cover within the following categories: canopy, sapling >1.5m, sapling <1.5m, shrub >1.5m, shrub <1.5m, *Rubus* spp., forb, fern, grass/sedge, coarse woody debris, leaf litter, bare ground, and *Vaccinium* spp./*Gaylussacia* spp. We recorded ocular tube readings at 10 locations per point: once at point center, once at 5 m along each of the 35 m transects, once at a random location along each 35 m transect, and lastly at the end of each of the 35 m transects. We calculated average percent cover for each strata by summing the total number of presences (1) and dividing that by that value by the total number of times a strata type could have been detected (total possible 10) at each plot. Lastly, we used a 10-factor wedge prism to estimate basal area at point center and at the end of each 35 m transect.

Avian surveys

We conducted point counts to 1) characterize breeding bird communities within each fenced and unfenced timber harvest, and 2) quantify patterns of settlement for a suite of focal species. Point count surveys are a simple and easily repeated method used to sample avian communities (Bibby et al. 2000; Ralph et al. 1995) and are often used when attempting to understand habitat effects on avian communities is the focus of an experiment (Bibby et al. 2000).

Assessing breeding bird community

The number of point counts surveyed within a given timber harvest depended on its size and shape. We ensured that all point count locations were at least 250 m apart and ≥ 50 m from a harvest edge, if this was not possible the survey was placed at the geographic center of the harvest. With these constraints in mind, we conducted 28 fenced and 27 unfenced surveys that were randomly placed within each fenced and unfenced harvest. These points were the same locations where the vegetation surveys were completed. We conducted two rounds of breeding season point counts; May 17 – 21 and May 28 – 31 2019 (Sorace et al. 2000). Visits were separated by approximately 7-14 days. All surveys began no earlier than 15 minutes post-sunrise and no later than four hours post-sunrise (Fuller & Langslow 1984; Lynch & Whigham 1984).

Upon arriving at each point, we waited 1 minute before initiating the survey to allow the birds time to settle down after potentially being disturbed (Bibby et al. 2000). Surveys were not conducted in unfavorable weather conditions, such as strong wind or rain (Holt et al. 2013). We recorded date, start time, cloud cover, temperature, and wind level (Johnson 1995) before starting each survey. During each 10-minute survey, we recorded the following for each

individual detected: species name, time interval within which the bird was first detected (0-2 minutes, 2-4 minutes, 4-6 minutes, 6-8 minutes, 8-10 minutes), sex (if possible), how each bird was detected (visual, song, call, or other), and distance from the observer when first detected (0-25 m, 26-50 m, 76-100 m, >100 m). Exact distances for the focal species was recorded.

Assessing timing of settlement

A recent study found that the degree of correlation between long-term population trends of forest bird and deer abundance estimates in Pennsylvania depended on each species primary nesting and foraging strata (Rushing et al. 2020). As such, we selected a suite of focal species that Rushing et al. 2020 found to be negatively associated with or impartial to deer abundance. The Chestnut-sided Warbler (*Setophaga pensylvanica*), Black-and-white Warbler (*Mniotilta varia*), Gray Catbird (*Dumetella carolinensis*), and Eastern Towhee (*Pipilo erythrophthalmus*), were predicted to be negatively associated species and the Common Yellowthroat (*Geothlypis trichas*) and Field Sparrow (*Spizella pusilla*) were predicted to be impartial species (Fig. 4.2).



Figure 4.2. The suite of focal forest bird species included in a study comparing avian communities in fenced and unfenced timber harvests in northcentral Pennsylvania: a) Chestnut-sided Warbler, b) Black-and-white-Warbler, c) Eastern Towhee, d) Field Sparrow, e) Gray Catbird, f) Common Yellowthroat.

We also conducted point count surveys prior to the breeding season surveys to compare territory settlement patterns between fenced and unfenced timber harvests. We surveyed the same point count locations and used the same protocol as the breeding season surveys. We conducted five rounds of pre-breeding season point count surveys (hereafter called settlement surveys) between 11 April – 8 May 2019: visit 1 occurred from April 11-14, visit 2 was April 17-18, visit 3 was April 23-25, visit 4 was May 1-2, and visit 5 was May 7-8. The five additional point count surveys provided a total of seven rounds of avian surveys for each point to assess territorial male settlement patterns and breeding territory densities.

Avian age structure

We used mist nets to target capture males of each focal species detected within fenced and unfenced timber harvests. We focused our capture efforts on Chestnut-sided Warblers and Common Yellowthroats because they showed a) different responses to deer (Rushing et al 2020), and b) were abundant across our sites. Mist netting occurred from 2 May – 7 June 2019. We placed one mist net (6-9 m long) at a location where a territorial focal male was detected. We tried to sample evenly across harvests working from one edge of the harvest to other. We used a recording of conspecific calls and songs to lure an individual into 1-2, 6-m long mist nets. We exerted the same amount of capture effort per pair of fenced-unfenced harvests. Upon the capture of each individual, we recorded the time, date, and coordinates of capture, species name, age, and sex of each individual. Additionally, we recorded the following morphometrics: tarsus length, wing cord, and total mass. To age each focal species, we used information from Peter Pyle's identification guide (Pyle 1997). All birds were aged as SY (second year) or ASY (after second year). SY birds are individuals that are within their second calendar year of life (their first breeding season) and ASY birds are at least within their third calendar year (≥ 2 breeding seasons). Each captured bird was also fitted with a uniquely numbered USGS aluminum leg band. Individuals were released at the location of capture and handling time was < 10 minutes.

Statistical Analyses

Paired vegetation and avian community comparisons

We analyzed all data using program R via RStudio (R Core Team 2019; RStudio Team 2019). For each measurement, we averaged plot-level values across sample locations for the fenced and unfenced sections of each site. We used a principle components analysis to compare relative amounts of cover types as measured using an ocular tube to identify which cover types differed most between sites (Kabacoff 2010; Jari et al. 2019). We also ran a series of paired t-tests comparing two measures of avian community and six measurements of vegetation. Our two measures of avian community were species richness and diversity (measured as effective species unit; Jost et al. 2006). For vegetation measures, we examined basal area (m^2/ha), horizontal and vertical vegetation density, along with three measures that were species specific: browse intensity, average stem height, and stem density. The species/genus groups differed for browse intensity due to the winter versus summer sampling and difference in sampling protocol. For browse intensity, we included species/genus groups with ≥ 30 stems in both treatment groups for all sites combined: birch (*Betula* spp.), black cherry (*Prunus serotina*), mountain laurel (*Kalmia latifolia*), oak (*Quercus* spp.), red maple (*Acer rubrum*), Rubus (*Rubus* spp.), sassafras (*Sassafras albidum*), serviceberry (*Amalanchier* spp.), and witch hazel (*Hammamelis virginiana*). For average height (m) and stem density (stems/ 5m^2), we included species with > 50 stems in both treatment groups for all sites combined: birch (*Betula* spp.), black cherry (*Prunus serotina*), mountain laurel, oak, red maple, Rubus (*Rubus* spp.), sassafras, serviceberry, and witch hazel. For each measure, we adjusted p-values based on the Holm-Bonferroni criteria to account for multiple testing among species (Holm 1979).

Avian density

We used hierarchical distance modeling to identify which variables were most strongly correlated with the density of our focal bird species (Kery & Royle 2016). We excluded the Eastern Towhee from these analyses due to lack of model convergence. To test for differences in species density (individuals/ha) between fenced and unfenced harvests and to identify which habitat variables influenced density, we used hierarchical distance modeling. Hierarchical

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 ≥ 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

model by assuring that the \hat{c} (variance inflation factor) values were less than 1. Values greater than 1 suggest overdispersion. Next, we ranked the models based on AIC_c values and identified variables that improved upon the null model by at least 2 Δ AIC_c. When more than one model improved upon the null model, we focused on the highest ranked model. When interpreting the results, we examined which visits experienced the largest increase in detections. We interpreted this time period to represent the greatest amount of territory establishment.

We used a chi-square contingency analysis to test for differences between age classes (Second Year [SY] and After Second Year [ASY]) of male Chestnut-sided Warbler and Common Yellowthroat in fenced and unfenced harvests. We only included males in our analysis due to small sample sizes for females. For each species, this resulted in four groups for comparison: ASY birds in fences, ASY birds in unfenced harvests, SY birds in fences, and SY birds in unfenced harvests.

Results

Browsing and vegetation surveys

We collected browsing data at 320 and 349 1 m radius plots in fenced and unfenced harvests, respectively. We had to move 1.9% (6/320) of plots in fenced and 16.3% (57/349) of plots in unfenced harvests due to lack of regeneration. Paired t-tests showed significant differences in the proportion of browsed stems between fenced and unfenced harvests. Specifically, oak spp., red maple, *Rubus*, and sassafras all had higher proportions of stems browsed in unfenced harvests compared to fenced harvests (**Table 4.2; Fig. 4.3**).

Table 4.2. Paired t-test results comparing the proportion of browsed stems by species between fenced and unfenced harvests. Browse surveys were conducted in March-April 2019 in northcentral Pennsylvania. Note: All p-values were Holm-adjusted.

Species Name	Number of stems		Percent of stems browsed		P – value
	Unfenced	Fenced	Unfenced	Fenced	
Oak (<i>Quercus</i> spp.)	396	442	90%	16%	< 0.001
Red Maple (<i>Acer rubrum</i>)	1140	738	94%	22%	< 0.001
Sassafras (<i>Sassafras albidum</i>)	144	478	80%	16%	< 0.001
Rubus (<i>Rubus</i> spp.)	450	1434	78%	6%	< 0.001
Witch Hazel (<i>Hamamelis virginiana</i>) ^a	208	219	58%	7%	0.05
Birch (<i>Betula</i> spp.)	78	80	42%	4%	0.4
Black Cherry (<i>Prunus serotina</i>)	34	93	47%	0%	0.4
Serviceberry (<i>Amelanchier</i> spp.)	58	76	81%	32%	0.4
Mountain Laurel (<i>Kalmia latifolia</i>)	168	170	8%	1%	0.4

^a p < 0.05 before Holm-adjustment



Figure 4.3. Examples of browsed woody stems found during browsing surveys in northcentral Pennsylvania.

Rubus was the only species for which stem density (stems/5m²) differed between unfenced and fenced harvests, with higher densities inside the later (**Table 4.3; Fig 4.11a**). Several species were significantly taller in fenced harvests compared to unfenced harvests: red maple and red oak, sassafras, serviceberry, and Rubus (**Figs 4.4 to 4.8, 4.11b; Table 4.3**). Chestnut oak and witch hazel heights did not differ between fenced and unfenced harvests (**Figs. 4.9 & 4.10**). Many of the species in the unfenced harvest were not (on average) tall enough to outreach deer browsing height (**Fig 4.11b**). Both horizontal (0.76 ± 0.06 in fenced vs. 0.54 ± 0.06 in unfenced; $p < 0.05$) and vertical (0.71 ± 0.05 in fenced vs. 0.42 ± 0.09 in unfenced; $p < 0.001$) vegetation density were significantly greater in fenced harvests.

Table 4.3. Total number of stems, mean height (m), and stem density (stems/5m²) for woody species in fenced and unfenced harvests in northcentral Pennsylvania. All p-values are Holm-adjusted.

Species	Total number of stems		Height (m) mean \pm SE ^a			Density (stems/5m ²) mean \pm SE		
	Fenced	Unfenced	Fenced	Unfenced	P value	Fenced	Unfenced	P value
Chestnut oak (<i>Quercus montana</i>)	135	113	2.1 \pm 0.31	1.4 \pm 0.39	0.35	1.4 \pm 0.74	1.1 \pm 0.54	1
Red maple (<i>Acer rubrum</i>)	682	617	1.7 \pm 0.29	1.0 \pm 0.22	0.05	7.6 \pm 1.4	6.3 \pm 1.4	1
Red oak (<i>Quercus rubra</i>)	86	76	2.1 \pm 0.37	0.93 \pm 0.20	0.05	1.1 \pm 0.44	1.2 \pm 0.73	1
Sassafras (<i>Sassafras albidum</i>)	187	60	2.0 \pm 0.33	0.83 \pm 0.34	0.04	2.2 \pm 0.63	0.52 \pm 0.2	0.08
Serviceberry (<i>Amalanchier spp.</i>)	90	59	1.5 \pm 0.11	0.66 \pm 0.05	0.02	0.82 \pm 0.32	0.53 \pm 0.22	1
Rubus (<i>Rubus spp.</i>)	1415	444	0.86 \pm 0.07	0.48 \pm 0.07	0.02	15.5 \pm 3.0	3.9 \pm 1.2	0.02
Witch hazel (<i>Hamamelis virginiana</i>)	176	128	1.7 \pm 0.11	1.5 \pm 0.15	0.28	1.3 \pm 0.6	2.1 \pm 1.4	1

^a Variable was log-transformed prior to statistical analysis

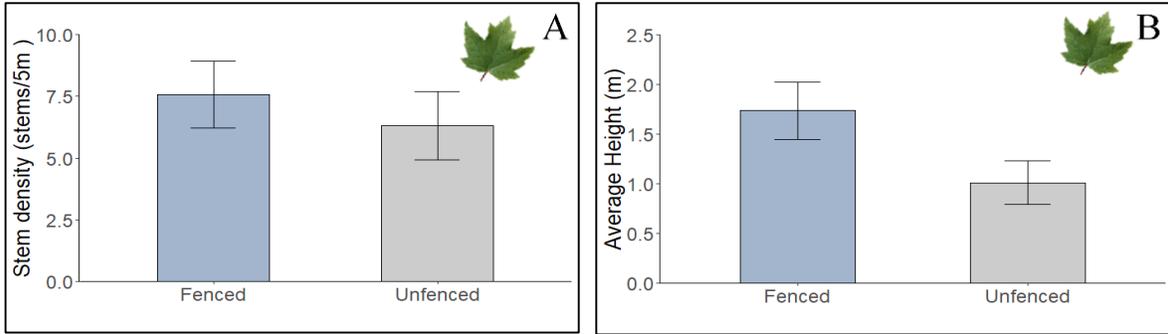


Figure 4.4. Average \pm SEM stem density (stems/5m²) (a) and average height (m) (b) for red maple (*Acer rubrum*) in fenced and unfenced harvests.

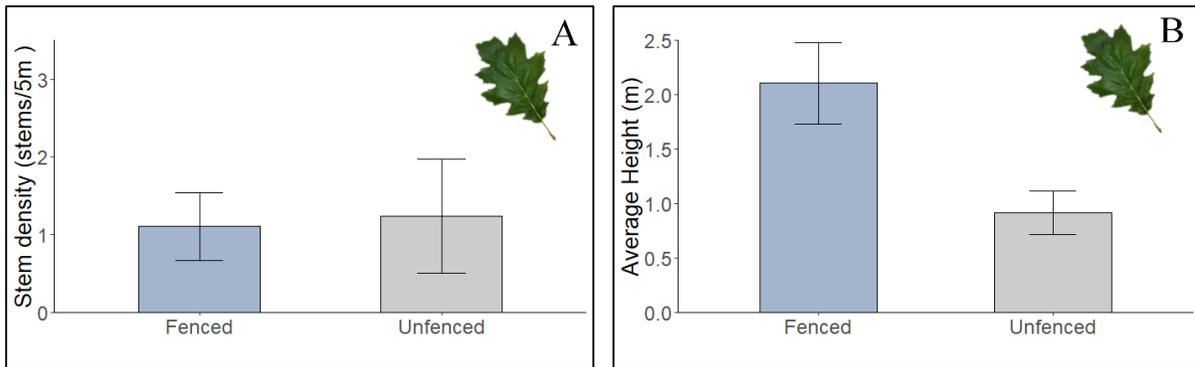


Figure 4.5. Average \pm SEM stem density (stems/5m²) (a) and average height (m) (b) for red oak (*Quercus rubra*) in fenced and unfenced harvests.

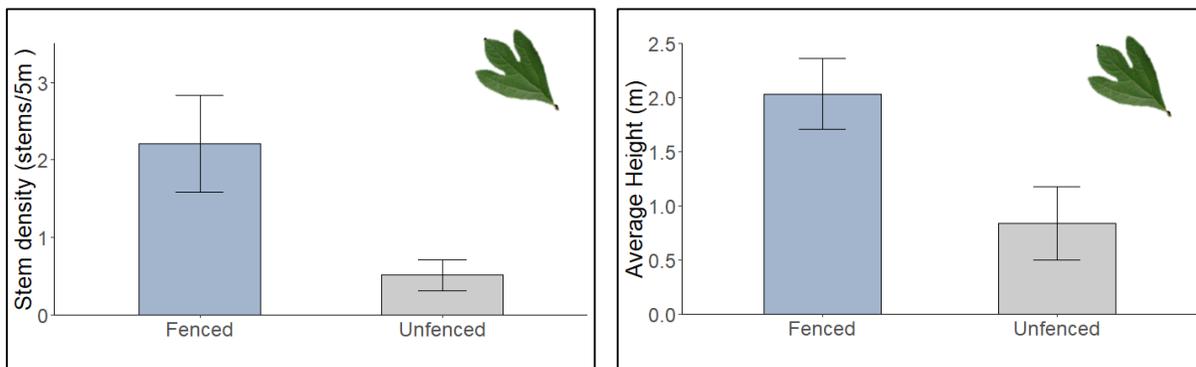


Figure 4.6. Average \pm SEM stem density (stems/5m²) (a) and average height (m) (b) for Sassafras (*Sassafras albidum*) in fenced and unfenced harvests.

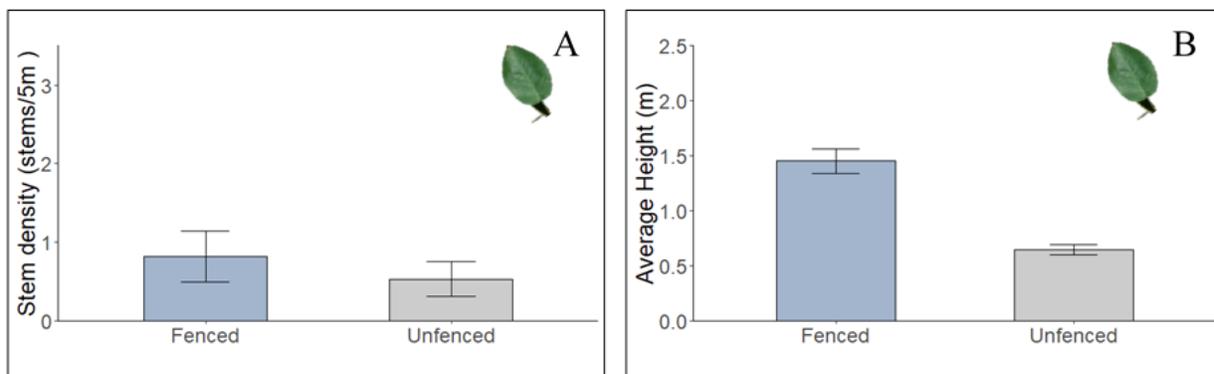


Figure 4.7. Average \pm SEM stem density (stems/5m²) (a) and average height (m) (b) for Serviceberry (*Amalanchier* spp.) in fenced and unfenced harvests.

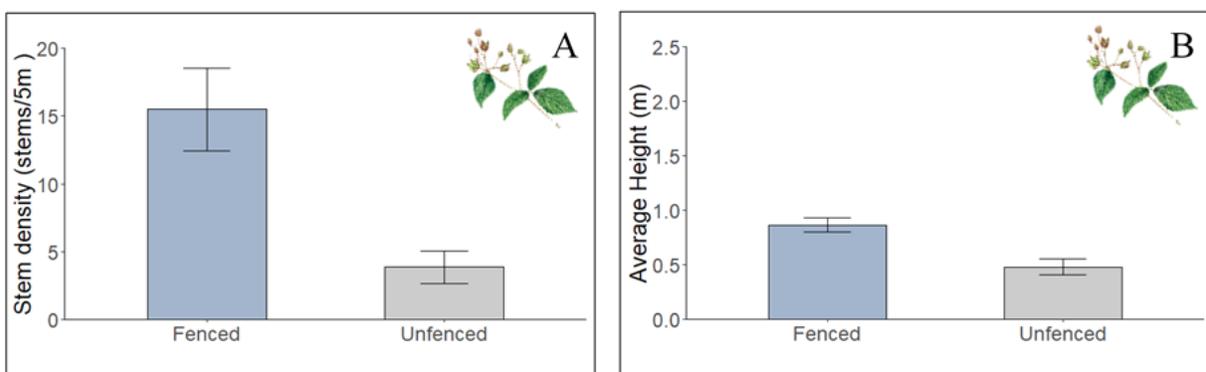


Figure 4.8. Average \pm SEM stem density (stems/5m²) (a) and average height (m) (b) for Rubus spp. in fenced and unfenced harvests.

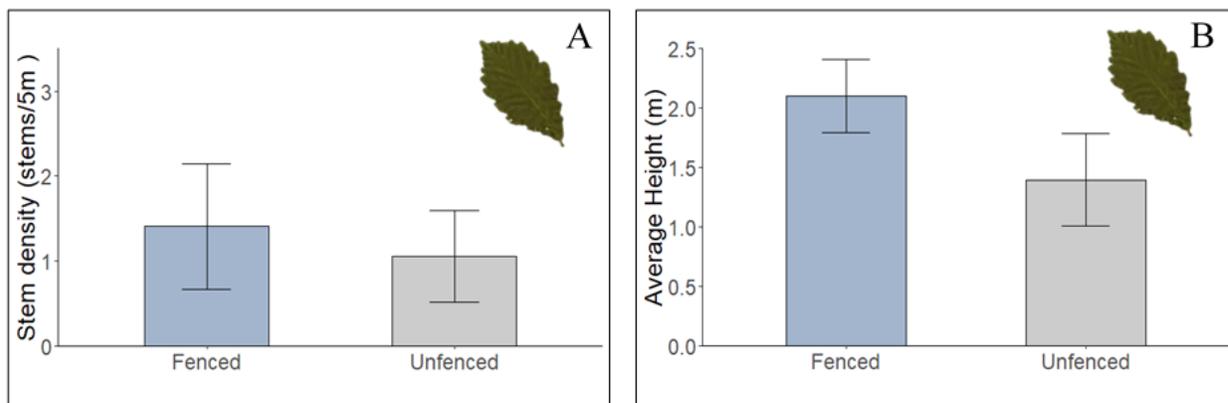


Figure 4.9. Average \pm SEM stem density (stems/5m²) (a) and average height (m) (b) for Chestnut oak (*Quercus montana*) in fenced and unfenced harvests.

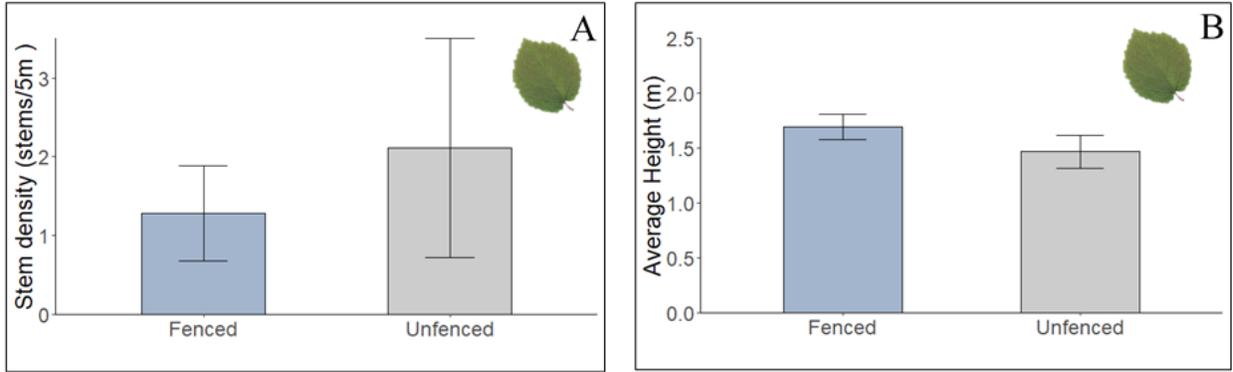


Figure 4.10. Average \pm SEM stem density (stems/5m²) (a) and average height (m) (b) for Witch hazel (*Hamamelis virginiana*) in fenced and unfenced harvests.

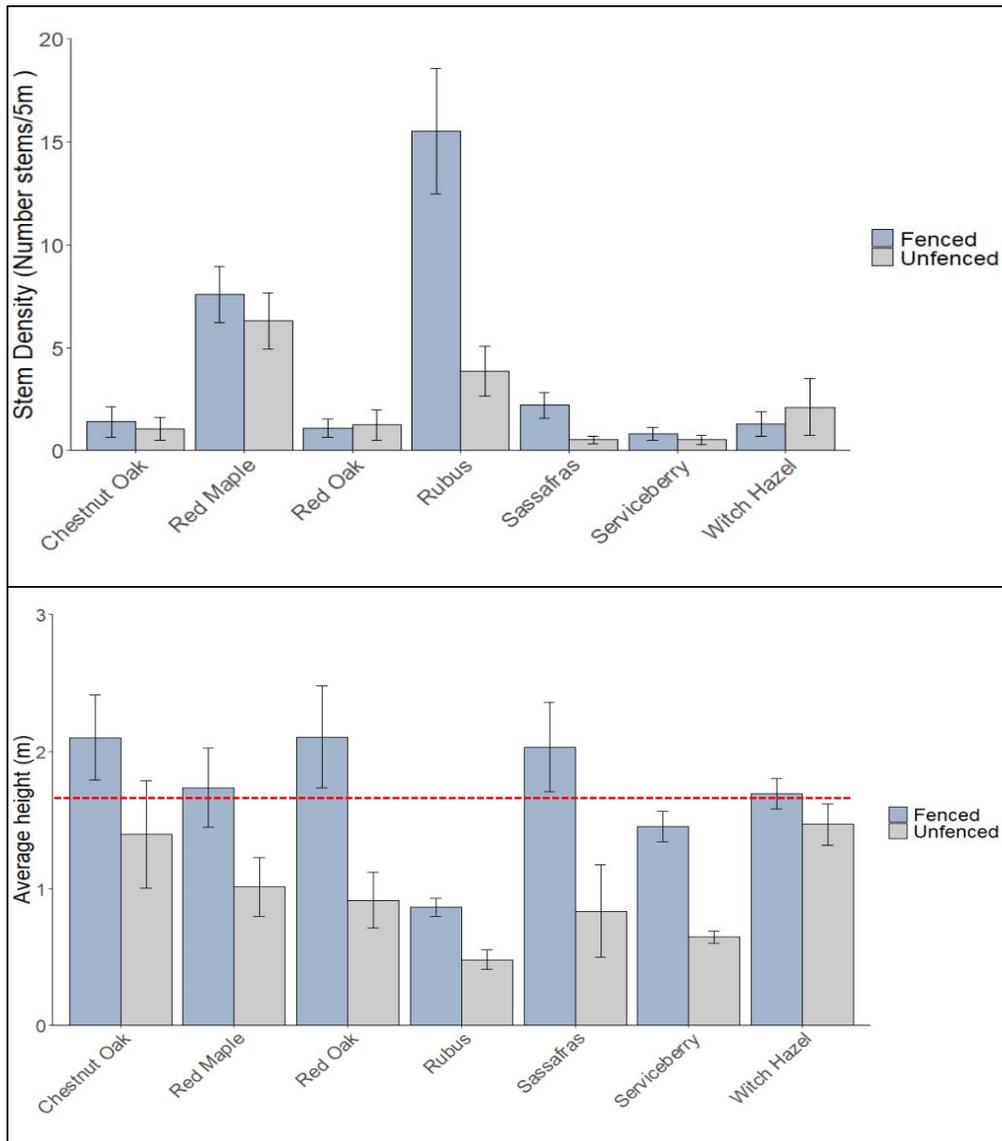


Figure 4.11. Comparison of a) average stem density (stem/5m²) and b) height (m) of woody species in fenced and unfenced timber harvests. The red dashed line represents typical height of browse line. Error bars represent standard error of the mean.

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**).

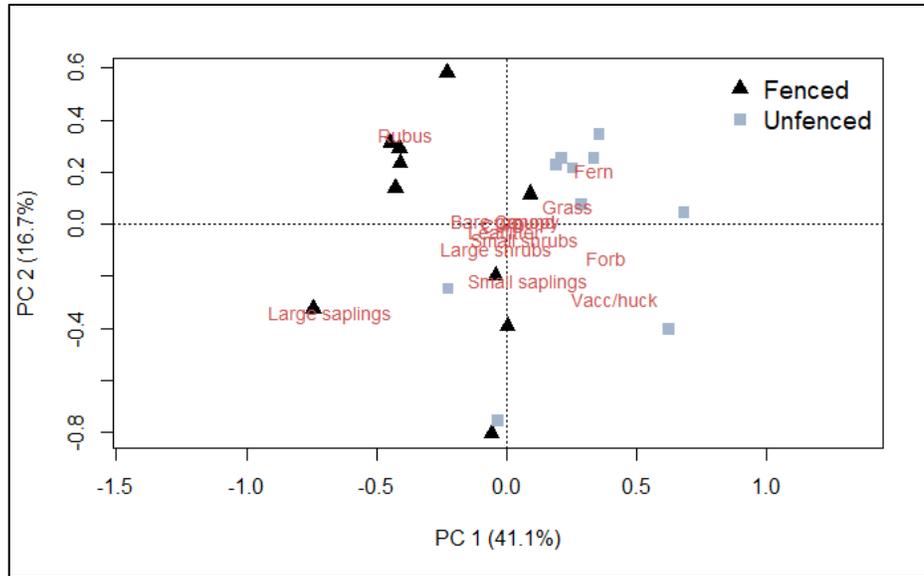


Figure 4.12. Principal components analysis for 13 vegetation cover variables measured in fenced and unfenced timber harvests in central Pennsylvania. Cover variables represent a suite of measures taken using an ocular tube and represent relative amounts of each habitat component.

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

Avian communities

Avian richness (12.4 ± 0.7 in fenced and 11.9 ± 0.6 in unfenced; $t = -0.5$, $df = 9$, $p = 0.6$), and effective species unit (9.3 ± 0.6 in fenced and 9.8 ± 0.5 in unfenced; $t = 0.7$, $df = 9$, $p = 0.4$) did not differ between fenced and unfenced harvests. We found $>1.5\text{m}$ saplings and vertical vegetation density were correlated (correlation = 0.8). Our hierarchical distance modeling results (**Appendix 4.2**) revealed that the Chestnut-sided Warbler had higher densities (individuals/ha) in fenced harvests and Gray Catbird densities were higher in areas with greater amounts of $>1.5\text{m}$ sapling cover (**Fig. 4.14; Table 4.5**). Rushing et al. (2020) predicted that Common Yellowthroat would have a neutral response to deer and our results support this prediction (Figure 14). Our results contrast with the predictions by Rushing et al. (2020) for Black-and-white Warbler and Field Sparrows (**Table 4.5**). Black-and-white Warbler density did not depend on treatment or related variables in our sample, although it was predicted to have a negative response (Table 5; Rushing et al. 2020). While Field Sparrows were predicted to be neutral, densities were found to be negatively associated with $>1.5\text{m}$ saplings (Table 5; Rushing et al. 2020).

Table 4.5. Hierarchical distance modeling results for our five focal forest bird species: Chestnut-sided Warbler, Common Yellowthroat, Eastern Towhee, Gray Catbird, Black-and-white Warbler, and Field Sparrow. Number of detections represent raw detections recorded during point count surveys. The top variable for predicting density represents our highest ranked model $> 2 \Delta AIC_c$ from the null. Predicted density is based off treatment (fenced of unfenced) model.

Species	Number of detections	Survey covariates affecting detection	Top variable for predicting density	Predicted density		Predicted response to deer (Rushing et al. 2020)
				Fenced	Unfenced	
Chestnut-sided Warbler (<i>Setophaga pensylvanica</i>)	350	None	Treatment	4.5 ± 0.46	3.0 ± 0.36	Negative
Common Yellowthroat (<i>Geothlypis trichas</i>)	209	Wind	Null	1.3 ± 0.18	1.4 ± 0.19	Neutral
Gray Catbird (<i>Dumetella carolinensis</i>)	94	None	$>1.5\text{ m}$ saplings	1.4 ± 0.71	1.0 ± 0.88	Negative
Black-and-white Warbler (<i>Mniotilta varia</i>)	101	None	Null	0.83 ± 0.35	0.74 ± 0.32	Negative
Field Sparrow (<i>Spizella pusilla</i>)	75	Time of day	$>1.5\text{ m}$ saplings	0.43 ± 0.14	0.64 ± 0.2	Neutral

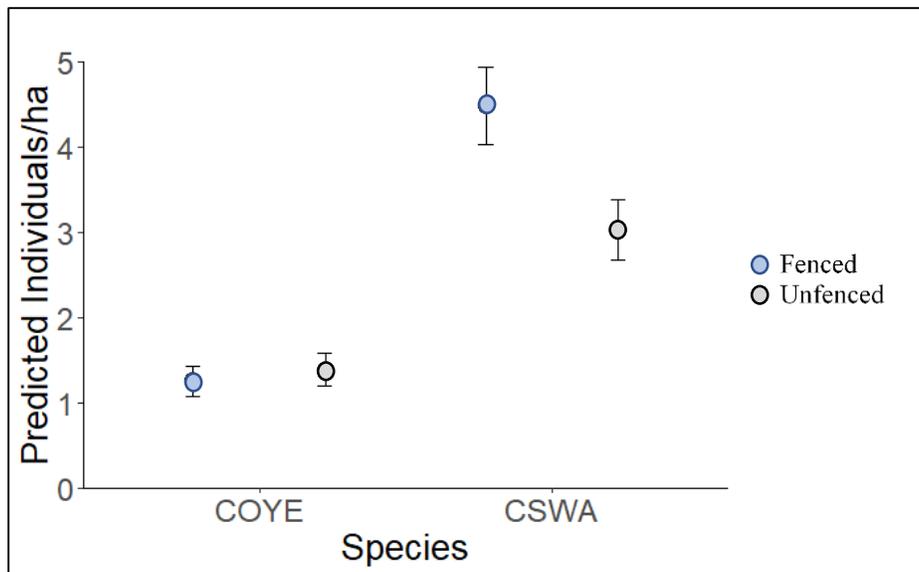


Figure 4.14. Predicted Chestnut-sided Warbler (CSWA) and Common Yellowthroat (COYE) density (individuals/ha) based on hierarchical distance modeling results between fenced (blue dots) and unfenced (grey dots) harvest. Error bars represent standard error of the mean

Avian Settlement surveys

Our territory settlement survey results suggested several of our focal bird species were influenced by either treatment, vertical vegetation density, or horizontal vegetation density (**Table 4.6; Appendix 4.3**). We assumed the greatest settlement period to be between the two visits with greatest difference in number of detections. For example, the Chestnut-sided Warbler’s settlement period was visit five, where the number of detections increased from 40 to 135. Chestnut-sided Warbler settlement was significantly influenced by increased horizontal vegetation cover. Eastern Towhee settlement was influenced by increased horizontal vegetation cover. Gray Catbird settlement was positively associated fenced sites. Field Sparrow settlement was positively associated with unfenced sites. In contrast, the Common Yellowthroat and Black-and-white Warbler settlement (like breeding survey results) showed no association with any of the habitat measurements we recorded.

Table 4.6. Highest ranked ($> 2 \Delta AIC$ from null model) generalized linear model for settlement survey results for six focal forest bird species by visit and number of detections. Chestnut-sided Warbler (CSWA), Common Yellowthroat (COYE), Eastern Towhee (EATO), Gray Catbird (GRCA), Black-and-white Warbler (BAWW), and Field Sparrow (FISP). Bolded values represent what we considered the primary settlement period based on largest increase in detections.

Species		Territory settlement visit with dates visited				
		1 April 11-14	2 April 17-18	3 April 23-25	4 May 1-2	5 May 7-8
CSWA	# Detections	0	0	0	40	135
	Highest Ranked Model	NA	NA	NA	Null	Horzveg (+)
COYE	# Detections	0	0	0	83	139
	Highest Ranked Model	NA	NA	NA	Null	Null
EATO	# Detections	43	186	204	199	155
	Highest Ranked Model	Vertveg (+)	Horzveg (+)	Null	Horzveg (+)	Null
GRCA	# Detections	0	0	0	45	63
	Highest Ranked Model	NA	NA	NA	Fenced (+)	Null
BAWW	# Detections	0	0	20	67	59
	Highest Ranked Model	NA	NA	Fenced (+)	Null	Null
FISP	# Detections	64	109	109	80	54
	Highest Ranked Model	Vertveg (+)	Fenced (-)	Null	Null	Horzveg (-)

Age class comparison

We captured and banded 147 male Common Yellowthroats and 156 male Chestnut-sided Warblers. We found no significant difference in age class distribution between fenced and unfenced harvests for either species (COYE; $X^2 = 0.07$, $df = 1$, $p = 0.8$ and CSWA; $X^2 = 1.2$, $df = 1$, $p = 0.3$) (Figs. 4.15 & 4.16).

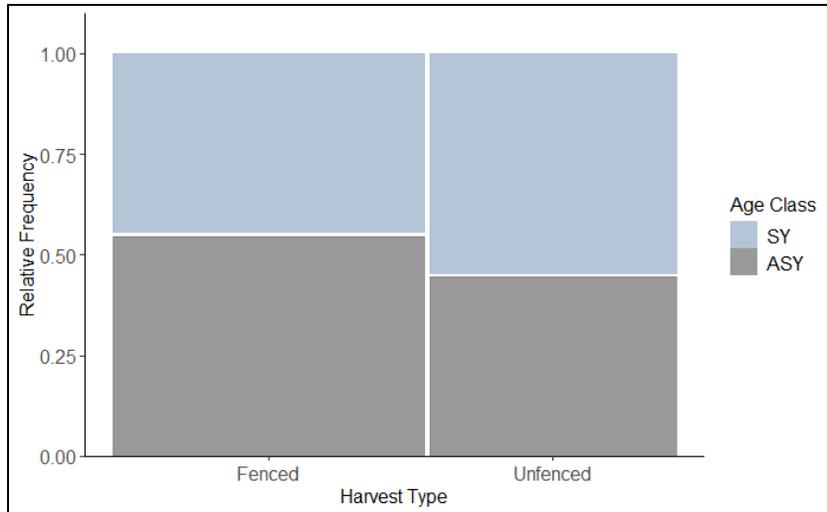


Figure 4.15. Chestnut-sided Warbler age class distribution (SY and ASY) by harvest type (Fenced or Unfenced) ($n = 156$ banded males).

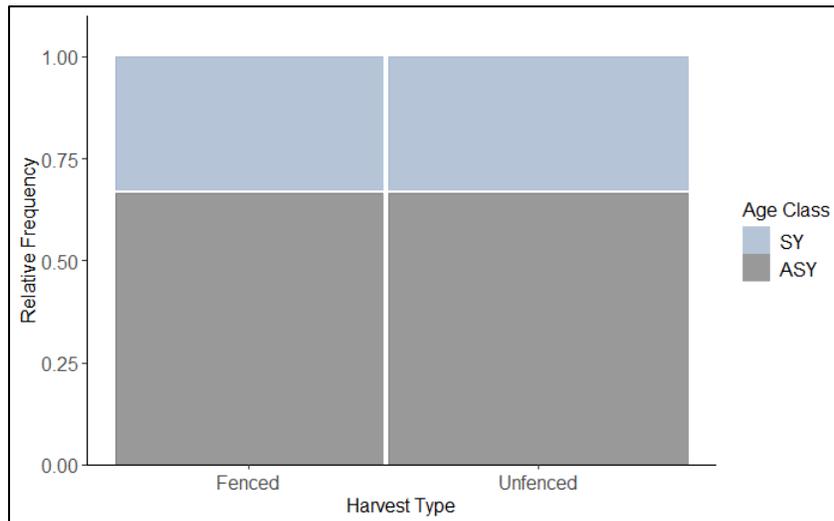


Figure 4.16. Common Yellowthroat age class distribution (SY and ASY) by harvest type (Fenced or Unfenced) ($n = 147$ banded males).

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

evidence that stem density for most species did not differ inside and outside fences, but average height did differ. Shorter average heights in unfenced harvests suggests those species stems will not outgrow the reach of deer. Our principal components analysis revealed fenced sites maintained more large saplings and *Rubus* cover, compared to unfenced sites. Whereas unfenced sites were characterized by having greater amounts of blueberry/huckleberry, forb, and fern cover.

We found no difference in overall avian community (richness and effective species unit) but did find higher Chestnut-sided Warbler densities in fenced versus unfenced harvests. Breeding densities for all but two species (Common Yellowthroat and Black-and-white Warbler) showed a significant effect of either horizontal vegetation density, vertical vegetation density, or treatment (fenced or unfenced). Thus, during territory settlement some species were selecting for habitats features that were influenced by deer exclusion fencing. Lastly, we found no difference in age structure for male Chestnut-sided Warbler or Common Yellowthroat captured in fenced and unfenced harvests. Initial findings from this study demonstrate clear impacts of deer browsing on vegetation structure in regeneration timber harvests, which will ultimately drive species composition of the future canopy. Moreover, vegetation features influenced by the fencing appeared to influence territory settlement and densities of some avian species. While deer exclusion fencing is a costly conservation/forest management practice, our data demonstrate that its use results in improved vegetation structure, increased heights of desired species of woody regeneration, and improved habitat conditions for several songbird species.

During the 2020 field season, we will conduct vegetation sampling at additional points within fenced and unfenced harvests to increase our woody species stem density and height dataset. We will also conduct at least five territory settlement surveys and two breeding season surveys at each of the 55 point count locations surveyed in 2019. Finally, if time allows and resources allow, we will attempt to recapture as many banded male Chestnut-sided Warbler and Common Yellowthroat as possible to assess annual site fidelity between fenced and unfenced harvests.

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Appendices

Appendix 4.1. Vegetation features measured in the ten paired fenced and unfenced harvests in northcentral, Pennsylvania.

Vegetation feature	Area measured along/within	Description
Proportion of browsed stems (0-1)	1m radius plots along transects	Proportion of woody stems browsed relative to all stems of that species recorded
Woody height (m)	3, 1x5m ² transects at each point	Woody height is represented by all woody species ≥ 0.1 meters measured with a meter stick
Woody density (stems/5m ²)	3, 1x5m ² transects at each point	Woody density is represented by number of woody stems of each species counted per transect (5m ²)
Horizontal vegetation density	Along three 35m transects at each point	Quantified using a vegetation profile board (Nudds 1977) that consists of 20, 20x20 cm squares. Density was measured by the proportion of squares $\geq 50\%$ obstructed by vegetation
Vertical vegetation density	Along three 35m transects at each point	Quantified using a densiometer (Lemmon 1956) that consists of 25 squares. Density was measured by the proportion of squares $\geq 50\%$ obstructed by vegetation
Vegetation cover	Along three 35m transects at each point	Quantified using a GRS densiometer (i.e. ocular tube). Averaged by the number of times a strata type was encountered by adding the number of presences (1) or absences (0) and dividing by the total number of times a strata type could have been detected (10) at each plot.
Basal area	35m transects at each point	Quantified residual basal area (RBA) using a 10-factor wedge prism

Appendix 4.2. Hierarchical distance model results for our focal forest bird species. Competing model list shows all models $> 2 \Delta AIC_c$ from null.

Species	Competing model list	AIC _c	ΔAIC_c	AIC _c Wt	Log-likelihood
Chestnut-sided Warbler	Treatment (fencing)	770.79	0.00	0.78	-381.00
	Global (Treatment+Horzveg+Rubus)	775.28	4.49	0.08	-380.77
	Horizontal vegetation density	775.31	4.52	0.08	-383.26
	Rubus cover	776.45	5.65	0.05	-383.82
Common Yellowthroat	Null	668.30	0.00	0.12	-328.54
Gray Catbird	Large saplings (>1.5m saplings)	447.64	0.00	0.55	-219.42
Black-and-white Warbler	Null	470.17	0.00	0.13	-230.69
Field Sparrow	Large saplings (>1.5m saplings)	373.58	0.00	0.86	-181.18

Appendix 4.3. Territory settlement results for our focal forest bird species. Territory settlement visit refers to the point count visit we considered having the highest territory establishment. Competing models are those $> 2 \Delta AIC_c$ from null. Global vegetation model combines both horizontal and vertical vegetation density measurements.

Species	Territory settlement visit	Model list	AIC _c	ΔAIC_c	AIC _c Wt	Log-likelihood
Chestnut-sided Warbler	5	Horzveg	175.17	0.00	0.35	-85.47
		Treatment	175.72	0.55	0.26	-85.74
		Global Veg	175.80	0.64	0.25	-85.67
		Vertveg	177.37	2.20	0.12	-86.57
		Null	180.56	5.39	0.02	-89.24
Common Yellowthroat	5	Vertveg	178.95	0.00	0.45	-87.36
		Null	180.64	1.68	0.19	-89.28
		Global Veg	181.11	2.15	0.15	-87.32
		Treatment	181.47	2.52	0.13	-88.62
		Horzveg	182.40	3.44	0.08	-89.08
Eastern Towhee	2	Horzveg	216.15	0.00	0.70	-105.96
		Global Veg	218.22	2.07	0.25	-105.88
		Treatment	221.38	5.23	0.05	-108.58
		Vertveg	226.99	10.84	0.00	-111.38
		Null	229.11	12.96	0.00	-113.52
Gray Catbird	4	Treatment	132.20	0.00	0.91	-63.98
		Vertveg	139.04	6.85	0.03	-67.41
		Null	139.48	7.29	0.02	-68.70
		Horzveg	139.87	7.68	0.02	-67.82
		Global Veg	140.75	8.55	0.01	-67.14
Black-and-white Warbler	4	Vertveg	133.59	0.00	0.26	-64.68
		Horzveg	133.64	0.05	0.25	-64.70
		Null	134.01	0.42	0.21	-65.97
		Treatment	134.73	1.14	0.15	-65.25
		Global Veg	134.81	1.22	0.14	-64.17
Field sparrow	2	Treatment	199.95	0.00	0.55	-97.86
		Null	202.67	2.72	0.14	-100.29
		Vertveg	202.92	2.97	0.13	-99.34
		Horzveg	203.01	3.06	0.12	-99.39
		Global Veg	204.48	4.53	0.06	-99.0