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Abundance Of Songbirds In Eastern Hemlock Stands Following Chemical Treatment For Hemlock Woolly Adelgid

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ABUNDANCE OF SONGBIRDS IN EASTERN HEMLOCK STANDS FOLLOWING CHEMICAL TREATMENT FOR HEMLOCK WOOLLY ADELGID

BY

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ABSTRACT

In the 1960's the invasive Hemlock Woolly Adelgid (*Adelges tsugae* [hereafter, HWA]) began to spread west across the hemlock stands of the Eastern U.S. killing a significant number of Eastern Hemlock (*Tsuga canadensis*). While chemical treatments, primarily with the active ingredient imidacloprid, have been implemented, their effects on hemlock dependent avian species are largely unknown. A 2009 study, which took place as HWA was beginning to invade Kentucky, identified six indicator species that were positively and negatively correlated with eastern hemlock stands throughout the Appalachian Mountain region of Kentucky. Our study repeated bird and vegetation surveys at the same 65 sites in 2018 (nine years later), to conduct a before-after-control-impact test of how the six avian indicator species and hemlock health have responded to chemical treatments. To better understand the mechanisms linking hemlock decline with changes in the bird community, the following three values were quantified: (1) the proportion of dead to live hemlock trees, (2) hemlock decline using an index based on canopy vigor, and (3) the importance value of hemlocks between years in treated vs. control sites. Generalized linear mixed models were used to ask if indicator bird species, abundances were related to percent dead hemlock, hemlock importance value, the hemlock decline index, chemical treatment, and year. Management area was included in all models as a random effect. Canonical Correspondence Analysis (CCA) was used to determine if the entire avian community had changed between years. We found an 11% increase in the percent dead hemlock across sites since 2009, regardless of chemical treatment. While hemlock canopies had higher vigor in treated sites, there was no significant difference in the hemlock decline index between treated and untreated sites. None of the six focal bird species showed a significant population response to chemical treatments (based on the interaction of treatment and year). The Black-throated Green Warbler (*Setophaga virens*) significantly declined between years across both treated and untreated sites. Although Acadian Flycatcher (*Empidonax virescens*) and Blue-headed vireo (*Vireo solitarius*) did not decline significantly over time, I found species association with hemlock and year to have a significant impact on the variation in total abundance of focal species. Red-eyed Vireo (*Vireo olivaceus*) and White-breasted Nuthatch (*Sitta carolinensis*) did not show a significant increase between years. However, Eastern Wood-Pewee (*Contopus virens*) significantly increased between 2009 and 2018. Our results suggest that while treatments have a positive effect on individual hemlocks, this effect is not carried over to the hemlock dependent avian species. This could be due to limitations in the effectiveness, as well as delays in implementation of chemical treatments within our sites. In fact, we may not see a significant decline in positive hemlock associates until some threshold of dead hemlock or poor hemlock health is met. This study suggests that more frequent and widespread treatment is needed to slow the decline of Kentucky's hemlock stands and protect hemlock-dependent avian species.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

Chapter I: Introduction

Invasive species are any species introduced in a manner that is not congruent with their natural migration or establishment process, which then have a negative impact on the ecosystem they subsequently inhabit (Sandlund et al. 2001). While the severity of the invasion of non-native species may vary depending on habitat conditions and species characteristics, it is widely accepted that they usually have negative impacts on wildlife and biodiversity (Sandlund et al. 2001). Once introduced, many invasive species contribute to the process of species homogenization due to their ability to outcompete specialist species (Clavero et al. 2009). Invasive invertebrate herbivores, specifically, exemplify these effects within forested ecosystems (Kenis et al. 2009). For example, some species defoliate and eventually kill trees, resulting in changes in light availability, hydrologic processes and forest tree composition (Kenis et al. 2009). Such drastic changes create an ecological cascade of negative impacts on the wildlife of forests (Kenis et al. 2009). The Hemlock Woolly Adelgid (*Adelges tsugae* [hereafter, HWA]) is one such invasive invertebrate that has altered Eastern Hemlock (*Tsuga canadensis*) ecosystems (Orwig and Foster 1998).

In the 1960's HWA began to spread west from Virginia across hemlock stands of eastern North America (Figure 1[1]). Originating in Asia, HWA is a part of the Order Hemiptera that is in part characterized by its sap-sucking mouthparts. In its native range its life cycle consists of two generations. The first generation (prodgrediens) occurs in the spring and are often referred to as crawlers (Cheah et al. 2004). This

[1] All Figures found in Appendix A

stage feeds and reproduces asexually on their hemlock host (Cheah et al. 2004). The second generation (sistens) is a winged generation that disperses in search of a spruce host to sexually reproduce on (Preisser et al. 2014). In North America, HWA only carry out asexual reproduction via, parthonogenesis (wherein only females occur) due to the lack of an appropriate Oicea sp. host (Preisser et al. 2014). Therefore, in North America, the crawler stage is a persistent force of the HWA spread. During this stage, HWA is wind dispersed or picked up by birds, humans and other mammals and brought to new areas (McClure 1990).

HWA targets the base of hemlock needles by latching on and sucking nutrients from xylem ray parenchyma cells (McClure 1987). HWA has a harmful spit that is transferred into the tree during feeding (McClure 1987). Both of these factors cause the tree to lose its existing needles and inhibit new growth. An infected tree can die in as little as one to four years depending on the size of the tree and the infestation severity, although some trees persist at low levels of vigor for decades after initial infestation (Benton et al. 2016a, Eschtruth et al. 2013). Hemlock regeneration only occurs through the seed bank, however, the seed bank is only viable for one growing season, making recovery of an infested stand difficult (Orwig and Foster 1998).

Disappearance of eastern hemlocks from the landscape has had lasting effects on the ecosystems it inhabits. The loss of adult eastern hemlocks increases woody debris and alters the age structure of forests stands (Orwig and Foster 1998). Further, with the reduction of shade from Hemlocks, deciduous species such as sweet birch (*Betula lenta*) and red maple (*Acer rubrum*) increased in some areas to dominate the understory (Orwig and Foster 1998). In southern regions, it predicted that evergreen, Rhododendron (*Rhododendron maximum*) will take over the understory, changing the vegetation and nutrient cycle (Horton et al. 2009). Together, these effects have led to a homogenization of the forest, and drastically changed the forest microenvironment and habitat suitability for wildlife species that prefer hemlock (Orwig and Foster 1998)

Forest stands dominated by Eastern Hemlock have higher avian diversity and richness than other forest types in the eastern United States (Howe & Mossman 1995). Their shade tolerance allows for retention of mid-story and lower branches providing complex vertical structure. Hemlock stands also provide foraging and nesting habitat for avian species that prefer hemlock dominated stands (Benzinger 1994). Acadian Flycatcher (*Empidonax virescens*), Black-throated Green Warbler (*Setophaga virens*) and Blue-headed Vireo (*Vireo solitarius*) are some of the species that prefer the complex vertical structure in hemlock stands as opposed to neighboring hardwood dominated stands (Howe & Mossman 1995, Tingley et al. 2002). The populations of these species and the overall avian diversity may be threatened by the negative effects of HWA. With a decrease in hemlock abundance, it is expected that the mid-story nesting bird species that prefer hemlock will decline, causing a decrease in avian species richness and overall forest health (Tingley et al. 2002, Toenies et al. 2018).

Biological and chemical methods have been developed to control HWA and thus mitigate the negative impacts on hemlocks. The most common biological control technique is the release of predatory beetles. Several species of beetles in the Coccinellidae and Derontidae family have been used for biological control of HWA, and some of these reduce HWA populations; however, because it can take some time for the population of beetles to become established, they are only effective in reducing the negative impacts on hemlocks if released in the early stages of infestation (Onken and Reardon 2011). Chemical treatments appear to be the most common and efficient way to control HWA. Chemical applications include soil or trunk injection treatments of a systemic insecticide, typically with the active ingredient Imidacloprid. Imidacloprid is a systemic pesticide that takes 4-12 weeks to be distributed throughout the tree depending on application methods and transpiration rates (Eisenback et al. 2014). Once absorbed, the chemical spreads throughout the tree to the xylem and is stored in xylem ray parenchyma cells for up to 7 years (Benton et al. 2016a; Eisenback et al. 2014). HWA that feed on the sap of treated trees die from neurotoxicity within 48 hours (Dilling et al. 2009). Reduced species richness has been observed in non-target detritivore and phytophagous guilds of insects, although most fatalities in non-target species result from foliar application as opposed to the more frequently used basal drench and soil injection methods (Dilling et al. 2009). Because Imidacloprid is water soluble, the amount of chemicals used in an area has a direct effect on the concentration of imidacloprid and its metabolites found downstream (Benton et al. 2016b). While at a low concentration imidacloprid poses no known threat to aquatic invertebrates. Even so, tree location, health and local climate should be considered for each treatment area before chemicals are applied to minimize non target impacts (Benton et al. 2016b and Eisenback et al. 2014).

Chemical and biological treatments have been applied to hemlock on private and public lands throughout the range of Eastern Hemlocks, including in southeastern Kentucky. Due to financial and logistical constraints, many treatments have been prioritized based on public use and the ecological significance of infested sites. In state-managed forests, treatments typically occur on a 3-5 year cycle, although practices vary depending on the specific management plan of each forest. While chemical treatments have been generally accepted as successful (Doccola et al. 2007), there are few follow-up studies on the ecological effects of imidacloprid.

A previous study compared avian communities in forest stands dominated by hemlocks to stands dominated by deciduous trees in southeastern Kentucky (Brown and Weinkam 2014). Brown and Weinkam (2014) identified four species expected to be negatively impacted by hemlock decline. Blue-headed Vireo, Acadian Flycatcher, Black-and-white Warbler (*Mniotilta varia*) and Black-throated Green Warbler demonstrated a positive association with hemlock trees, and were predicted to decline with the expected loss of hemlocks. In contrast, the authors found 10 species would be positively impacted by hemlock mortality including, Mourning Dove (*Zenaida Macroura*), Red-bellied Woodpecker (*Melanerpes carolinus*), Red-eyed Vireo (*Vireo olivaceus*), White-breasted Nuthatch (*Sitta carolinensis*), and Eastern Wood-Pewee (*Contopus virens*); all negatively associated with hemlocks and predicted to increase in abundance (Brown and Weinkam 2014). Notably, this study took place prior to heavy infestation of HWA, and only provided predictions of avian responses to the expected hemlock decline.

Objectives

My study is a follow up on previous research of Brown and Weinkam (2014). I revisited the hemlock dominated study areas throughout southeastern Kentucky to determine how the abundance of avian focal species has changed over time. Based on the strength of their association with hemlock, sample size, and results of other studies that suggested associations with hemlock, six focal species were selected to compare their abundances between the 2009 study and 2018. The positive associates with hemlock I assessed were Blue-headed Vireo, Acadian Flycatcher, and Black-throated Green Warbler. The Black-and-white Warbler was not included in this study because of conflicting reports in the literature of whether it is positively or negatively associated with hemlock (Keller 2004). The negative associates I assessed were, Red-eyed Vireo (*Vireo olivaceus*), White-breasted Nuthatch (*Sitta carolinensis*), and Eastern Wood-Pewee (*Contopus virens*). Many of the study sites I evaluated have been managed with imidacloprid. Thus, I was poised to conduct a before-after-control-impact test of how focal avian species and hemlock trees responded to chemical treatments by comparing the abundance of focal species between 2009 and 2018 in treated and untreated areas. To better understand the mechanisms linking hemlock decline and changes in the bird community, the following predictor variables were quantified: (1) the proportion of dead to live hemlock trees, (2) hemlock decline index, and (3) the importance value of hemlocks between years in treated vs. control sites.

My study contributes to the understanding of the long-term effectiveness of chemical treatments for invasive insects in regards to forest health and avian populations, and thus provides insight on how current and future management strategies could be modified to benefit forest health and wildlife communities.

Chapter II: Methods

Study Area

This study was limited to eastern Kentucky, where hemlock is found in pockets along the Appalachian range on the eastern side of the state. This portion of the state includes the Western Allegheny Plateau, Central and Southwestern Appalachians ecoregions (Woods et al. 2002). In Brown and Weinkam (2014), 72 hemlock dominated sites were established in 10 management areas within these ecoregions (Figure 2). Brown and Weinkam (2014) included an additional 35 sites dominated by hemlock that had been surveyed annually by the Kentucky Department of Fish and Wildlife but those surveys were discontinued by 2010. Brown and Weinkam (2014) also included 16 sites that are surveyed annually by the US Forest Service, but use different vegetation and avian survey protocols. Due to these discrepancies we only attempted to survey the 72 sites. The sites were categorized as hemlock stands if Eastern Hemlock was observed as being one of the top three most abundant overstory trees. In 2009, each site was marked with a numbered metal tag, a white PVC pipe sunk at least 0.5 m into the ground and protruding approximately 0.2 m, flagged with colored tape, and coordinates were recorded with a GPS. At each site, a 7854 m2 circular plot (50-m radius) was established from the PVC-marked center point. For this study, which took place from 2017 to 2018, I located 67 sites out of the originally established 72 sites (Table 1[2]). Avian population data were collected for all 67 sites, but vegetation

[2] All tables found in appendix B

data were only collected for 65 sites, therefore all data analyses were based on those 65 sites for which both vegetation and avian data were available.

The history of chemical and biological treatments for HWA was obtained for each management area from land managers and through personal observations at each site. Chemical treatments for HWA were conducted in 9 of the 10 management areas. At each of these 10 areas, treated hemlock trees were marked at the time of pesticide application with a spray-painted bright colored circular dot, approximately 5 cm in diameter. Between the years 2008–2010, 18,908 trees were treated at Bad Branch State Nature Preserve (Kyle Napier, Kentucky Office of State Nature Preserve). A total of 10,332 of these trees were then retreated between the years 2012–2014. Similarly, Blanton Forest State Nature Preserve had 10,290 trees treated between 2008–2010, with 5,020 trees retreated during 2012–2014 (Kyle Napier, Kentucky Office of State Nature Preserve). Cranks Creek Wildlife Management Area had no record of treatment (Kyle Napier, Kentucky Office of State Nature Preserve). In 2011–2012 and 2013–2014, land managers at Cumberland Falls State Resort Park treated a total of 11,240 hemlock trees, but I was unable to determine if any of these were repeated treatments of the same trees over multiple years (Abe Nielsen, Kentucky Department of Parks). Within Cumberland Gap National Historical Park, six of eight sites were chemically treated. Of these six sites, three were treated once and three were treated twice between the years of 2008 and 2016 (Jennifer Beeler, U.S. National Park Service). The two remaining sites within this management area were treated with beetles as a biological control in 2009, but not with chemicals. It is unknown to what degree of establishment these beetles achieved, and for the purpose of this study these sites were considered untreated. At Laurel Gorge Wildlife Management Area, 50.59 hectares were treated in 2013–2014, but the total number of trees treated was not recorded (Scott Freidhof, Kentucky Department of Fish and Wildlife Resources). It is unknown how many trees were treated or retreated at Lilley Cornett Woods because they did not keep useful records. Based on observations and conversations with land managers we know treatments were carried out in three separate years and thousands of trees across all of the old-growth forest were treated. Pine Mountain Settlement School carried out treatments in 2009 (2314 trees inoculated), 2011 (293 trees inoculated), 2013 (2748 trees inoculated) and 2015 (955 trees inoculated). With the records provided I was unable to determine how many, if any, trees were retreated within these years (William Field, Pine Mountain Settlement School). Pine Mountain State Resort Park treated 2,557 trees in 2017–2018 (Kyle Napier, Office of State Nature Preserve; Joe Hacker, Kentucky Division of Forestry). Stone Mountain Wildlife Management Area had 421 trees treated in 2017–2018 (Kyle Napier, Office of State Nature Preserve). The number of trees and years of treatment varied greatly among the 10 management areas and 65 sites. For this study, I considered a site within a management area to be chemically treated for HWA if at least a single counted hemlock within 50 m of the site center had a treatment mark (i.e., spray paint dot). At almost all sites with evidence of treatment, most hemlocks surrounding the study site had been treated as well. In all, 36 sites were categorized as being chemically treated and 31 sites were untreated.

Vegetation Sampling

In 2017, vegetation surveys were carried out by Rachel Miller, an NSF Research Experiences for Undergraduates student, using standard forestry techniques at 32 of the 65 sites, primarily at the Cumberland Falls State Resort Park, Bad Branch State Nature Preserve, Lilley Cornett Woods, Pine Mountain Settlement School, and Blanton Forest State Nature Preserve management areas (Miller 2017). In 2018 I carried out vegetation surveys at the other 33 sites included in this study. The vegetation data collected included diameter at breast height (DBH) of trees at the study site, the number of chemically treated hemlocks and the canopy vigor of each hemlock. These values were used to calculate the hemlock decline index, hemlock dominance, hemlock density, importance value and percent dead hemlock at each site. In 2009, vegetation surveys followed the same protocol with the exception that hemlock canopy vigor was not assessed, although at the time HWA had only recently invaded and most trees appeared healthy, even if some had evidence of early stages of infestation (Brown and Weinkam 2014).

DBH was measured for all trees determined to be “in” the plot area based on sighting with a 10-factor prism from the center of the plot (Wensel et al. 1980). DBH values were used to calculate the total basal area of all hemlocks in each plot. The basal area of hemlocks was then divided by the total basal area of the site to determine the proportion of hemlock basal area that was composed of hemlock in a site (i.e., hemlock dominance). The number of hemlock trees was divided by the total number of trees to determine the relative density of hemlocks for each site. The hemlock dominance and relative density values were summed to calculate the importance value of hemlocks for each site.

It was noted whether each individual hemlock was treated or not within a 50-m radius of the plot center during the vegetation surveys. All hemlocks that were measured for DBH also had the canopy vigor assessed on a scale of 1-5 (1 = 76–100%, 2 = 51–75%, 3 = 26–50%, 4 = 1–25%, 5 = Dead) (Smith 2006). A hemlock decline index (HDI) was calculated by summing the basal area of hemlocks with canopy vigor ratings of 3-5 and dividing by the total hemlock basal area for each site. The percentage of dead hemlocks was calculated by dividing the number of hemlocks with a canopy vigor of 5 (i.e., dead) by the total number of hemlocks in the plot.

Focal Avian Species

Blue-headed Vireos are found in mixed deciduous-coniferous forests throughout the Appalachians (Hudman and Chandler 2002). Foraging in a wide variety of live trees, these vireos use sally-strikes to catch moths and other flying insects (Holmes and Robinson 1981). Blue-headed Vireos also hop through mid-level branches gleaning caterpillars and other larvae (Holmes and Robinson 1981). Blue-headed Vireos prefer to nest on the lower branches of young conifers such as hemlocks and spruce (Bent et al. 1950), placing their nest 2-5 meters above the ground (Burleigh 1958). Out of the four species identified as being positively associated with hemlocks by Brown and Weinkam (2014), Blue-headed Vireos showed the strongest association with hemlock. Though their association with hemlock is clear, the effects that hemlock decline has on their abundance and behavior is unknown.

Acadian Flycatchers are typically found in mature forests and forested wetlands. Throughout Appalachia, Acadian flycatchers are associated with damp hemlock dominated forests (Allen et al. 2009). Although Acadian Flycatchers are generalists with regard to the trees they forage in, they typically forage from the shrub layer and higher using a sit-and-wait approach (Farnsworth and Lebbin 2004, Guilfoyle et al. 2002). This species eats a wide variety of insects from moths to beetles (Farnsworth and Lebbin 2004). Acadian Flycatchers often use Eastern Hemlock for nesting sites, and utilize their needles as nesting material (Allen et al. 2009). Allen and colleagues (2009), found that breeding pairs of Acadian flycatchers decreased by 70% in heavily HWA infested Appalachian highland forest sites in Pennsylvania and New Jersey. This suggests that hemlock decline could severely impact their breeding habitat and populations.

Black-throated Green Warblers inhabit mixed deciduous-coniferous forests and are sensitive to fragmentation (Hagan et al. 1996). Their diet consists mainly of caterpillars gleaned from conifers (Macarthur 1958). They prefer to nest in hemlock stands or forests dominated by other conifer species, and often abandons heavily infested patches of hemlock, thus preferring to use uninfested patches (Tingley et al. 2002). Black-throated Green Warblers populations are expected to decline in response to HWA (Tingley et al. 2002, Brown and Weinkam 2014).

In the Appalachian Mountains, Red-eyed Vireos are found in deciduous forests that are dominated by Red Maple (*Acer rubrum*) and species of red oak (*Quercus coccinea*, *Q. rubra, and Q. velutina*) (Hudman & Chandler 2002). They typically forage in the canopy of trees and nest on medium to low branches of a variety of deciduous trees (Lawrence 1953). They have been observed in higher numbers in areas with high density of dead hemlocks, suggesting they may respond positively to the decline of hemlock (Tingley et al. 2002).

White-breasted Nuthatch prefer mature deciduous and coniferous forests. They typically forage on trunks and large branches probing the bark for larval arthropods. Being secondary cavity nesters, they utilize natural or avian made cavities. Though they prefer to nest in live trees they have been observed nesting in snags as well (Stauffer and Best 1980). They have been found in higher abundance in forest stands with relatively high hemlock mortality though the cause of this association is unclear (Tingley et al. 2002).

Eastern Wood-Pewees prefer forests with an open understory or edge habitat (Hespenheide 1971). When foraging they perch on dead branches (mean height = 11 m) and sally for flying insects (Johnston 1971, Fitzpatrick 1980). They prefer to nest on dry ridge tops and avoid wet valleys or north facing slopes where hemlocks tend to occur (Newell and Rodewald 2011). This may be one of the factors driving their negative association with hemlock.

Bird Abundance Sampling

Bird surveys were conducted following the protocol used in 2009 by Brown and Weinkam (2014). For optimal detection all point count surveys were conducted between May 10 to June 30, 2018, following the main period of spring migration and while breeding birds are still actively nesting and singing. Each site was originally established with a minimum of 250 m of separation between sites to decrease the probability of recounting individual birds. Point counts began as early as 30 min before sunrise and were completed prior to 3 hrs after sunrise. At the first point of the day, temperature, cloud cover and wind speed were recorded. Wind speed was assessed on the Beaufort scale (0–6) (Hamel et al. 1996). If it was drizzling or the wind was above a value of 4, then point counts were discontinued. Each point count lasted for 15 min total with playback of indicator species broadcast through a speaker during the last 5 min. For initial detections of each individual bird, the species, the time elapsed since the beginning of the survey, and the location relative to the site center were recorded on a spot map marked with circular distance intervals of 10 m to a maximum distance of 50 m.

Data analysis

Linear mixed models were used to test how the response variables hemlock importance value and hemlock decline index differed between sites with and without chemical treatment, and between survey years. After testing the assumptions of normal distributions and homogeneous variances, I used the function “lmer” from package “lme4” in R 3.6.1. to analyze a single hemlock decline index model and a series of importance value models. Importance value was analyzed using a series of candidate models including a combination of the explanatory variables treatment and year (Table 2). Management area was included in all models as a random effect since multiple sites occurred within each area. The four candidate models were compared by AIC_c score, with the lowest AIC_c score indicating the best model, and any model within a ΔAIC_c of 2.0 was included as a competing model. Model averaging was used to obtain parameter estimates and 95% confidence intervals for multiple competing models ($\Delta AIC_c < 2.0$) using package "MuMIn" and function model.avg(). If the null model was included among top competing models the results were considered not significant. The hemlock decline index model only included treatment as an explanatory variable due to the absence of 2009 canopy vigor data. It also contained management area as a random effect. To determine the statistical significance of the model an ANOVA was used to compare the global model and the null model using the function "anova" in base R for each vegetation response variable respectively and reported with a chi-square statistic.

Percent dead hemlock was analyzed using a Generalized Linear Mixed Model (GLMM) due to the non-normal nature of the data and inclusion of management area as a random effect. The global model contained percent dead hemlock as the response variable and the main effects and interaction between year and chemical treatment as the explanatory variable. The data were grouped by the number of hemlock trees recorded to be dead or alive at each site. A binomial error distribution was used and the fit was assessed with a chi-square goodness of fit test. To determine the statistical significance of the explanatory variables, model selection was used following the same candidate models and methods as the importance value analysis.

Due to avian abundance data having a heterogeneous distribution, GLMMs were constructed to analyze the abundance of each species (Bolker 2009). It is also important to note that the abundance data were not adjusted for detection probabilities. For each bird species, the global models based on three different distributions, Poisson, Negative Binomial and Zero-inflated Poisson, were compared via AIC_c scores using package "AICcmodavg" and function "aictab." I used package "lme4" with the functions "glmer" and "glmer.nb" to run and compare the GLMMs with Poisson and Negative Binomial error distributions. The model with the lowest AIC_c score was then compared to the model with a Zero-inflated Poisson distribution as calculated using package and function "glmmTMB". A chi-squared goodness-of-fit test was run on the global model with the lowest AIC_c score to determine if the distribution selected was a good fit. The best distribution was then applied to the set of candidate models in the final model selection process for each respective species. Each of the avian species positively associated with hemlock (Blue-headed Vireo, Acadian Flycatcher, Black-throated Green Warbler) and negatively associated with hemlock (Red-eyed Vireo and White-breasted Nuthatch) had the same series candidate models (Table 3). Management area was included in each model as a random effect to account for variation in hemlock density and the spatial clustering of sites. The 11 candidate models were compared by AIC_c score, with the lowest AIC_c score indicating the best model, and any model within a ΔAIC_c of 2.0 was included as a competing model. Model averaging was used to obtain parameter estimates and 95% confidence intervals for multiple competing models ($\Delta AIC_c < 2.0$) using package "MuMIn" and function model.avg(). If the null model was included among top competing models the results were considered not significant.

Eastern Wood-pewee abundance was analyzed differently than the other avian species. Although Eastern Wood-Pewee was shown to have a negative association with hemlock by Brown and Wienkham (2014), they had zero detections at the 65 (of 123 total) sites that were used again in the current study. This resulted in the 2009 data that are used in this study as having zero observations for this species. Therefore we chose to group the data for this species by management area and used the number of sites with and without detections (i.e. frequency) as the response variable. We then used a GLMM with year as the explanatory variable in the model. We were unable to include chemical treatment as a variable because most management areas included treated and untreated sites. A Binomial error distribution was fit to the model and the goodness-of-fit was determined using a chi-squared goodness-of-fit test. To determine the statistical significance of the model likelihood ratio test was used following the same methods as stated in the vegetation analysis.

In addition to the analysis on individual species, I performed another analysis for which the abundance was summed across species based on their positive and negative association with hemlock. The grouped abundance was then used as the response variable and allowed inclusion of association as a categorical variable in the GLMM models. A set of candidate models was created with chemical treatment, percent dead hemlock, year, importance value of hemlock and avian association with hemlock as explanatory variables (Table 4). The same distribution selection, model selection and model averaging procedures that were used for individual species were followed using the grouped data.

To better understand the effects of environmental variables on the entire avian communities in hemlock stands, we conducted a Canonical Correspondence Analysis (CCA) in R using package Vegan. The analysis included all species that were observed more than twice. To reduce clutter in the figure for this analysis, only the 65% most abundant species were included in the plot and no transformation was applied. The environmental variables (explanatory variables) included management area, importance value of hemlock, percent dead hemlock, treatment and year. Using the function “ordistep” we selected the best environmental variables to explain the avian community. The function carries out a permutation test and selects the significant variables based on the F-statistic (Borcard and Legendre 2011). A separate permutation test was conducted with function “anova.cca” to identify the significance of the overall CCA model, environmental variables and CCA axes.

Chapter III: Results

The hemlock importance values in treated and untreated sites between years were not significantly different (Figure 3A), although the pattern among years and treatments appears to show the importance value in untreated hemlock declined over time and in comparison to treated hemlocks. The hemlock decline index global model, which included treatment, was also not significant (χ^2 = 1.30, df = 1, P = 0.25, Figure 3B). However, when looking at overall hemlock canopy vigor, one of the components of the hemlock decline index, there were fewer hemlocks with poor canopy vigor ratings in treated sites than untreated sites (Figure 4).

The chi-squared goodness-of-fit test showed that a binomial distribution was a good fit for the global model evaluating percent dead hemlock (χ^2 = 0.78, df = 125). In 2009, out of 431 hemlock trees assessed, only 3% were dead, while in 2018, 11% of 485 trees were dead. The proportion of dead hemlocks increased between 2009 and 2018 regardless of chemical treatment (Figure 5). The most plausible model contained the explanatory variable year, and explained 64% of the variation in percent dead hemlock (Table 5). There was only one competing model (ΔAIC_c = 1.11) which contained the interaction between treatment and year and explained 36% of variation in percent dead hemlock. When averaged, the coefficient confidence intervals of these models showed that year had the strongest positive effect on percent dead hemlock (Table 6).

For all avian focal species abundance, the global model with a poisson distribution had the lowest AIC_c score compared to models of other distributions. The lowest ΔAIC_c was 2.23 and the average w_i across species was 0.76 providing support for poisson being the best distribution. The chi-squared goodness-of-fit tests indicated that poisson distribution was a good fit for the global model of each avian species (Table 7). The null model was either the most plausible or included as a competing model for the abundance of Acadian Flycatchers and Blue-headed Vireos (Figure 6). Black-throated Green Warbler however showed a statistically significant decline in abundance between years (Figure 7). Between 2009 (n = 87) to 2018 (n = 47) Black-throated Green Warblers declined by 46%. The most plausible model for Black-throated Green Warbler included only the effect of year, and explained 49% of the variation in Black-throated Green Warbler abundance (Table 8). The only competing model (ΔAIC_c = 1.75) contained percent dead hemlock and the additive effect of year. This model explained 20% of variation in abundance. When averaged, the coefficient confidence intervals of these models showed that year had the strongest negative effect on the abundance of Black-throated Green Warblers (Table 9). The null model was among the top models for all of the negative-associated avian species indicating no significant difference in abundance based on year or treatment (Figure 8).

In 2009, zero Eastern Wood-Pewees were observed at our 65 sites, however, in 2018, five were observed. Out of the four sites at which they were detected, three were chemically treated. The binomial distribution was a good fit for the global model analyzing Eastern Wood-pewee abundance (χ^2 = 0.98, df = 17). When comparing the Eastern Wood-Pewee global model, which only included year, to the null model we found that it was significant (χ^2 = 6.10, df = 1, P = 0.01), indicating a significant increase in abundance of Eastern Wood-Pewees between years (Figure 9).

The Poisson distribution had the best fit for the global model used to assess the effect of avian association on abundance (w_i = 0.59). The most plausible model for total bird abundance included the interaction between avian association and year, which explained 95% of the variation in abundance. The closest competing model had a ΔAIC_c of 7.67 and a weight of 0.02. Overall, positive associates declined by 34% between 2009 (n = 193) and 2018 (n = 128, Figure 10). Negative associates increased by 39% between 2009 (n = 41) and 2018 (n = 57).

Of the 58 avian species detected at the 65 sites in 2009 and 2018, 45 were observed at least twice and therefore fit the criteria for inclusion in the CCA analysis. The selection process used to determine what environmental variables best explained the community data in the CCA showed that management area and year should be included in the model. The final model explained 7% of the variation and was statistically significant (F = 2.14, P = 0.001). Out of 13 canonical axes, the top four were significant, and, axes 1 (F = 5.24, P = 0.001) and 2 (F = 4.31, P = 0.001) explained 27.1% and 22.2% percent of the variation, respectively (Table 10). Our model showed that the avian community was significantly different between management area (F = 2.19, P < 0.001) and year (F = 3.54, P < 0.001). Black-throated Green Warbler and Blue-headed Vireo appeared to be correlated with the year 2009, and less so with the year 2018, indicating a decline (Figure 11). Eastern Wood-Pewee was closely correlated with the year 2018.

Chapter IV: Discussion

Hemlock woolly adelgid is not the first invasive insect to degrade the complex forest structure and the avian community in Appalachia. Invasive insects have been recorded affecting breeding bird densities, nest success, and overall community composition due to their rapid spread and impacts on forest health (Rebenold et al. 1998, Koenig et al. 2013). For example, the Balsam Woolly Adelgid (*Adelges piceae*) invaded the Appalachian region in 1955, it completely eliminated Fraser fir (*Abies fraseri*), and led to breeding bird densities declining by 50% over a 20 year period in some areas (Rebenold et al. 1998). Additionally, in 2007 the Emerald Ash Borer (*Agrilus planpipennis*) began to spread into the Appalachian region, reaching Kentucky in 2009 (Haack et al. 2015). The spread was facilitated by the movement of infested firewood and nursery stock. Since its initial introduction, millions of ash trees have died. Statewide quarantines and extensive research into possible chemical and biological treatments have occurred, and state and federal management plans have been implemented (Buck 2015). However, any management efforts have largely been ineffective except for protecting small numbers of trees with expensive treatments that will likely need to be conducted annually or biennially decades into the future (Mulroy et al. 2019). In both cases, forests have been drastically changed and sensitive forest specialists have been nearly extirpated due to the decline of these specific forest types (Rabenold et al. 1998). Species that associate with Eastern Hemlock are now also vulnerable as hemlock woolly adelgid moves through the Appalachian region. The invasion of these pests have biological and economic costs and understanding those impacts is essential to mitigating the damage the next invasive species may cause.

My study is the first to investigate the effects of imidacloprid treated areas on the hemlock associated bird community across one ecoregion over multiple years. However, Falcone et al. (2010) attempted to identify the effects of treatment on avian populations, but their follow up study took place only a year after chemical treatment. As a result they found no difference in avian density between chemically treated and untreated sites. For other invasive pests, like the gypsy moth (*Lymantria dispar*), studies were carried out early in the biological control treatment process to determine possible side effects of non-target species die off on avian species richness, abundance and diversity (Cooper et al. 1990, Moulding 1976 and Strazanac and Butler 2005). In fact, most studies on the topic are focused on how decline of non-target species affects other trophic levels. Few to no studies have attempted to link the improvement of tree health due to chemical treatment to changes in avian community composition through time.

We expected to see a significant decrease in hemlock health at sites that were not chemically treated between 2009 and 2018, and therefore an increase in percent dead hemlock within those sites. While we did find an increase in percent dead hemlock over the 10 years between sampling, and many of the remaining hemlocks are of poor vigor, hemlock continues to have a high importance value in all of the stands used in this study. Surprisingly, we found no evidence that treatment reduced hemlock mortality. Other research has shown significant differences in adelgid infestation due to the number of years post-treatment and initial severity of infestation at the time of treatment (Benton et al. 2016a). Benton et al. (2016a) found that while imidacloprid was effective in suppressing HWA for 4-7 years, there was a high degree of variability in retention within hemlocks among sites. It has also been shown that increases in canopy vigor and overall hemlock health after treatment depends on the initial health of the tree when treatment occurs (Webb et al. 2003). The management areas within our study had a wide range of years since treatment last occurred, as well as variation in the frequency of treatments during the study period (i.e., some areas were treated more than once), and when treatments first occurred relative to the onset of infestation. Stone Mountain State Resort Park and Pine Mountain State Resort Park were treated for the first time in 2018, and two other sites were treated once, but not retreated during the following 4-5 years. One site, Lilley Cornett Woods did not keep accurate records of treatment, which complicates our ability to understand the efficacy of treatments, such as in this study. Some of the sites (i.e., Pine Mountain State Resort Park) that were considered treated in the analysis may have been treated too late into infestation to fully arrest the effects of HWA. This lack of consistency in years treated and delayed treatments may explain the lack of statistical significance of chemical treatment in the models and the increase in percent dead hemlock at both treated and untreated areas. Although it's likely that hemlock mortality would be more widespread in the absence of treatments, our data suggest that declines are occurring regardless of treatment.

Although ground cover data within our stands is available, it was not analyzed as part of my thesis. Based on our personal observations, the understory within treated and untreated sites had substantial cover. In 2018, rhododendron was the species most often reported as dominant in the understory of our stands. Our observational data supports the hypothesis that rhododendron could become a dominant species crowding out understory competitors and taking over what were once vertically unique hemlock stands if this decline continues (Ford et al. 2012, Mulroy et al. 2019). This anecdotal observation suggests what the plant community may look like if hemlock decline continues unchecked; further investigation is certainly merited

All but one avian species positively associated with hemlock showed non-significant declines, and the variation in abundance was not explained by the environmental variables included in the models. This lack of significant decline may be due to plasticity in habitat use by the species (Brown and Weinkam 2014). Though Blue-headed Vireo and Acadian Flycatcher have been shown to be positive hemlock associates, there may be environmental cues other than hemlock health affecting their habitat selection, such as relative elevation or proximity to streams. In some parts of the Acadian Flycatchers range, it is thought that they may be able to shift their habitat use as hemlocks decline or remain in an area longer due to the better health of hemlocks near streams (Allen et al 2009, Becker et al. 2008). This ability to use neighboring habitats as hemlock stands degrade may explain the lack of decline in abundance for these positive associates. The CCA also indicated that there was significant variation in the bird community between management areas. This indicated that we were correct in accounting for management area as a random effect within our models. In addition to landscape factors, such as relative elevation, it is possible there were other random effects, such as climate related variables, that we were not able to account for that added to the variation in our data set and made it difficult to detect differences in abundance between years in treated and untreated areas.

The Black-throated Green Warbler, which has been described as more of a hemlock obligate than the other positive associates (Benzinger et al 1994), was the only species to significantly decline in abundance between years. Brown and Weinkam (2014) found Black-throated Green Warblers to be the most strongly associated with Eastern Hemlock within the same management areas studied here and suggested that they required a high density of hemlock to maintain their populations. This suggests that hemlock decline may lead to more rapid and larger decreases in abundance for this species compared to the other bird species that are positive associates. The analysis evaluating hemlock association supports this expectation, indicating that the interaction between association and year contributed to the variation in abundance. Also the CCA indicated that there was a significant change in the avian community between years further supporting that there has been a shift in avian presence within our study area. Toenies et al. (2018) suggested that positive associates may remain on the landscape until there is an absence of stands with high basal area in low hemlock health or low basal area in high hemlock health. In fact, we may not see a significant decline in positive hemlock associates until some threshold of dead hemlock, health and abundance is met (Toenies et al. 2018).

Red-eyed Vireo and White-breasted Nuthatch were expected to show significant increases in abundance between years, however, this was not observed. Red-eyed vireo typically nest in deciduous trees with substantial canopy cover (Lawrence 1995). While it is thought that they may move into dying hemlock stands due to the potential increase in deciduous vegetation, this change in the forest canopy can take years or even decades to reach the high percentage of overstory canopy cover preferred by Red-eyed Vireos. This delayed effect of the turnover from hemlock to deciduous forest, plus the short-term increase in open sky when hemlock mortality increases, may slow Red-eyed Vireo from moving into the area for some time. White-breasted Nuthatch was expected to move into areas with high hemlock mortality due to stand composition changes and new nesting opportunities. They are considered to be generalists though they are typically found in deciduous stands with an open understory. As previously stated, some stands within our study area did not yet have deciduous species as dominant overstory trees. In fact, most of our stands still had hemlock as the midstory dominant tree. Furthermore, in areas with high recruitment of deciduous trees, the understory may be more dense than typically preferred by White-Breasted Nuthatch. In about half of our sites, the understory was estimated to have 50-100% cover. Being secondary cavity nesters they are expected to move into an area with increased snags, although they often prefer nesting in live trees. Once a pair begins to breed, however, their cryptic behavior may have reduced the probability of detection within our hemlock sites (Tingley 2002 and Kilham 1972). This could cause our study to underreport the abundance of White-breasted nuthatch within our sites.

The Eastern Wood-Pewee was the only negative associate to show a significant increase between years. Though the number of individuals recorded was low (n = 5) this is still an important finding due to the well-recorded negative association these birds have with Eastern Hemlock (Becker et al. 2008, Tingley et al. 2002, Brown and Weinkam 2014). Not only do Eastern Wood-Pewees not frequent hemlock stands, but they have been shown to move into sites as hemlock mortality increases (Tingly et al. 2002). Becker et al. (2008) suggest that Eastern Wood-pewees become more prominent in decaying stands due to the increase in open foraging habitat and dead perching branches they rely on within the midstory. While my data could not be used to analyze the relationship that Eastern Wood-Pewee have with the percent dead hemlock, their absence at our 65 sites in 2009 and presence in 2018 suggests that the avian community is beginning to shift due to the decline of hemlock stands.

Across North America there have been mass declines in breeding birds since the 1970's (Rosenberg et al. 2019). Tyrant flycatchers and New World warblers were two of the 38 families reported to have experienced significant declines while Vireos showed an increase in abundance (Rosenberg et al. 2019). Within the Appalachian region, analysis of the North American Breeding Bird Survey indicates the Eastern Wood-Pewee and Acadian Flycatcher as decreasing in abundance, whereas, Blue-headed Vireo, Red-eyed Vireo, Black-throated Green Warblers and White-breasted Nuthatch have increased in the Appalachian region (Pardieck et al. 2019). Within

Kentucky specifically these trends have held true for all focal species addressed in this study except for the Red-eyed Vireo and Acadian Flycatcher (Pardieck et al. 2019). The Acadian Flycatcher appears to be steadily increasing within Kentucky, whereas the Red-eyed Vireo's trend has been flat (Pardieck et al. 2019). Within our sites we saw a significant increase in Eastern Wood-Pewee abundance and a decrease in the abundance of Black-throated Green Warbler. This does not fit the regional or state trends, which suggests the local trends are directly related to the decline of hemlock, as caused by HWA, and not to broader scale changes. Though the other hemlock associates did not react significantly, all except the White-breasted Nuthatch shifted in abundance against their regional and state trends, and all changed in the direction expected due to the decline of hemlock. Since 2009, White-breasted Nuthatch appears to have increased at a state and regional level, if this trend continues we expect there to be a significant increase in their abundance at our sites within the next decade (Pardieck et al. 2019). With the regional and state trends contradicting the local trends shown in this study, we suggest a further need for long-term, monitoring-based research of bird communities in hemlock forest in the Appalachian region.

There were limitations to the study design that may influence the results, such as a limited number of study sites, and a single avian survey per site in each year. Brown and Weinkam (2014) established 72 sites in 2009, and were able to use data from an additional 51 sites, but only 65 were surveyed in 2018. This relatively small number of sites increases the risk of Type II error, and underreporting rare species (Thompson and Schwalbach 1995). By investigating species that were likely to be in the study area due to habitat association these biases may have been mitigated (Thompson and Schwalbach 1995). Only one round of point counts were conducted in 2009 and 2018 giving a single data point for each year for our comparison. Though our study had replication throughout our management areas, multiple rounds of point counts may have aided in untangling the local shifts in abundance from nationwide trends. Avian populations oscillate yearly due to density dependent factors and environmental stochasticity (Sæther et al 2016). With a single point count to represent each time period, it increases the risk that differences in abundance were found simply by chance due to yearly differences in abundance. It also limits our ability to predict specific abundances for our focal species.

Cold temperatures late winter can increase yearly HWA mortality and even slow the spread of infestation in the northern part of the species distribution (Parker et al 1998, Paradis et al. 2008). States with an average winter temperature ≤ 5°C are thought to have complete HWA mortality (Paradis et al 2008). If low temperature duration or severity kills off 91% of HWA, these conditions can keep it from spreading (Paradis et al 2008). These cold climates create a northern line of defense against further spread of HWA and decline of hemlock. With the increase in temperatures due to climate change, however, it is expected that the range of HWA will expand. Conservative models suggested that in the coming decade climate change will cause the temperature-protected portion of the distribution of hemlock to be reduced to half its current size implying that HWA will more easily expand its range (Paradis et al 2008). Though climactic events were not taken into account within this study, unusually cold temperatures in some years may have indirectly impacted the lack of significance in our predictor variables (i.e., chemical treatments, and year in some analyses) by diminishing infestation impacts in subsequent years. Specifically, in February and March of 2015, Kentucky experienced record low temperatures that may have increased overwintering mortality of HWA (Kentucky Mesonet http;//www.kymesonet.org/historical_data.php). This could be one reason Kentucky has not experienced the same level of hemlock mortality as in more southern neighboring areas.

It is clear that the Appalachian landscape has changed due to HWA. Throughout its range, Eastern Hemlock has already been degraded to astonishing levels in regards to stand health and density. With warming temperatures HWA is expected to spread further across the distribution of hemlocks. It has been predicted that Eastern Hemlock would be functionally extinct within its range by 2030 (Spaulding and Rieske 2010). Mulroy et al. (2019) supported this prediction by reporting almost complete mortality of all hemlock trees in the Great Smoky Mountains that were not treated. Our study also supports the argument for an eventual functional extinction of hemlock by showing an increase in percent dead hemlock on the landscape regardless of treatment. While it is promising that we found no significant decline in two positive avian associates of hemlock, our results do show significant declines for one species and when the three positive associate species were combined there is an overall decline over time, thus there appears to be a trend towards declining populations of species positively associated with hemlock that may become exacerbated in the future if treatment regimens do not become more robust and standardized. It is also important to note that the rate of decline may increase if a potential threshold for loss of hemlock is reached.

Based on my research, if the current pattern of hemlock decline continues in southeastern Kentucky then we can expect functional extinction within another decade. Eleven percent of hemlock within our sites have died in recent years, with many others in poor health, and the bird community has changed with declines of species that positively associate with hemlock. The loss of Kentucky's hemlock habitat will contribute to homogenizing the forested landscape and the loss of biodiversity across a large number of taxonomic groups in the Appalachian region (Fassler et al. 2019, Brooks 2001, Ingwell et al. 2012). Although our results suggest only minimal benefits to birds of chemical treatments for HWA, because of the likelihood of threshold effects, land managers should focus on early intervention treatments before severe infestation. This will allow for an expansion in the number of trees that can be chemically treated before HWA takes over a site. By maintaining the health of Kentucky’s hemlock stands, an important part of breeding bird habitat can be preserved and we can combat the further decline of North American avian biodiversity.

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APPENDICES

Appendix A: Tables

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Table 1. Number of sites established in 2009 and the number of those sites revisited in 2018. Out of the total number of sites relocated the number of sites considered to be chemically treated is indicated in the treated column.

Management Area	Number of Sites Established in 2009	Number of Sites Found in 2018	Number of Sites Treated
Bad Branch State Nature Preserve	12	10	9
Blanton Forest State Nature Preserve	8	7	4
Cranks Creek Wildlife Management Area	5	5	0
Cumberland Falls State Resort Park	8	8	1
Cumberland Gap National Historical Park	8	8	6
Laurel Gorge Wildlife Management Area	6	6	3
Lilley Cornett Woods	8	7	5
Pine Mountain Settlement School	6	5	2
Pine Mountain State Resort Park	5	5	2
Stone Mountain Wildlife Management Area	6	6	4
Total	72	67	36

Table 2. Models used to analyze importance value in a linear model candidate selection process. The model displayed in bold is the global model.

Models
~ 1
~ Treatment * Year
~ Treatment
~ Year

Table 3. Models used for each avian species of interest in the GLMM candidate model selection process. The model displayed in bold is the global model.

Models
~ 1
~ Treatment * Year + IV + Percent Dead Hemlock
~ Treatment * Year + IV
~ Treatment * Year + Percent Dead Hemlock
~ Treatment * Year
~ IV + Percent Dead Hemlock
~ Year + Percent Dead Hemlock
~ Treatment + Percent Dead Hemlock
~ Treatment + Year
~ Treatment
~ Year
~ IV
~ Percent Dead Hemlock

Table 4. Candidate models for GLMM selection process including association as an explanatory variable. The bold model is the global model.

Models
~ 1
~ Treatment * Year + IV + Percent Dead Hemlock + Association
~ Treatment * Year + Percent Dead Hemlock + Association
~ Treatment * Year + IV + Association
~ Treatment * Year + Association
~ Year* Association
~ IV + Percent Dead Hemlock + Association
~ Year + Percent Dead Hemlock + Association
~ Treatment + Percent Dead Hemlock + Association
~ Treatment + Year + Association
~ Treatment + Association
~ Year + Association
~ IV + Association
~ Percent Dead Hemlock + Association
~ Association

Table 5. Top models for percent dead hemlock model selection. K = number of parameters; AIC_c= Akaike's information criterion corrected for finite sample sizes; ΔAIC_c = Delta AIC_c; w_i= Akaike's weight

Model	K	AIC_c	ΔAIC_c	w_i
~ Year	3	218.57	0	0.64
~ Treatment * Year	5	219.68	1.11	0.36

Table 6. Individual explanatory variable coefficient estimates, standard errors, and 95% Confidence Interval from model averaging top models in percent dead hemlock candidate model selection analysis.

		95% Confidence Interval	
Variable	Estimate (Standard Error)	Lower	Upper
Intercept	-4.221 (0.564)	-5.335	-3.108
Year	1.815 (0.452)	-0.922	2.708
Treatment	0.341 (0.592)	-0.327	2.196
Treatment * Year	-0.209 (0.503)	-1.954	0.806

Table 7. Descriptive statistics for Chi-squared goodness-of-fit test (χ^2) determining fit of Poisson distribution to the global model for analysis abundance of Acadian Flycatcher (ACFL), Blue-headed Vireo (BHVI), Black-throated Green Warbler (BTNW), White-breasted Nuthatch (WBNU), Red-eyed Vireo (REVI).

	ACFL	BHVI	BTNW	WBNU	REVI
χ2	0.974	0.821	0.985	0.999	0.891
Residual degrees of freedom	121	121	121	121	121
Residual deviance	92.654	106.618	89.748	56.049	102.212

Table 8. Top models for Black-throated-green warbler GLMM model selection. K = number of parameters; AIC_c= Akaike's information criterion corrected for finite sample sizes; ΔAIC_c = Delta AIC_c; w_i= Akaike's weight.

Model	K	AIC_c	ΔAIC_c	w_i
~ Year	3	300.97	0	0.49
~ Percent Dead Hemlock + Year	4	302.73	1.75	0.2

Table 9. Individual explanatory variable coefficient estimates, standard errors, and 95% Confidence Interval from model averaging top models in Black-throated Green warbler candidate model selection analysis.

		95% Confidence Interval	
Variable	Estimate (Standard Error)	Lower	Upper
Intercept	0.161 (0.167)	-0.168	0.489
Year	-0.684 (0.189)	-1.077	-0.292
Percent Dead Hemlock	0.122 (0.414)	-0.867	1.695

Table 10. Five out of 13 conical axis from canonical correlation analysis (CCA) on 45 avian species and three vegetation variables for 65 sites in Kentucky.

Axis	Df	F	P-value	Eigenvalue
1	1	5.24	0.001	0.271
2	1	4.31	0.001	0.222
3	1	3.09	0.001	0.160
4	1	2.01	0.059	0.104
5	1	1.57	0.445	0.081

Appendix B: Figures

Appendix B: Figures

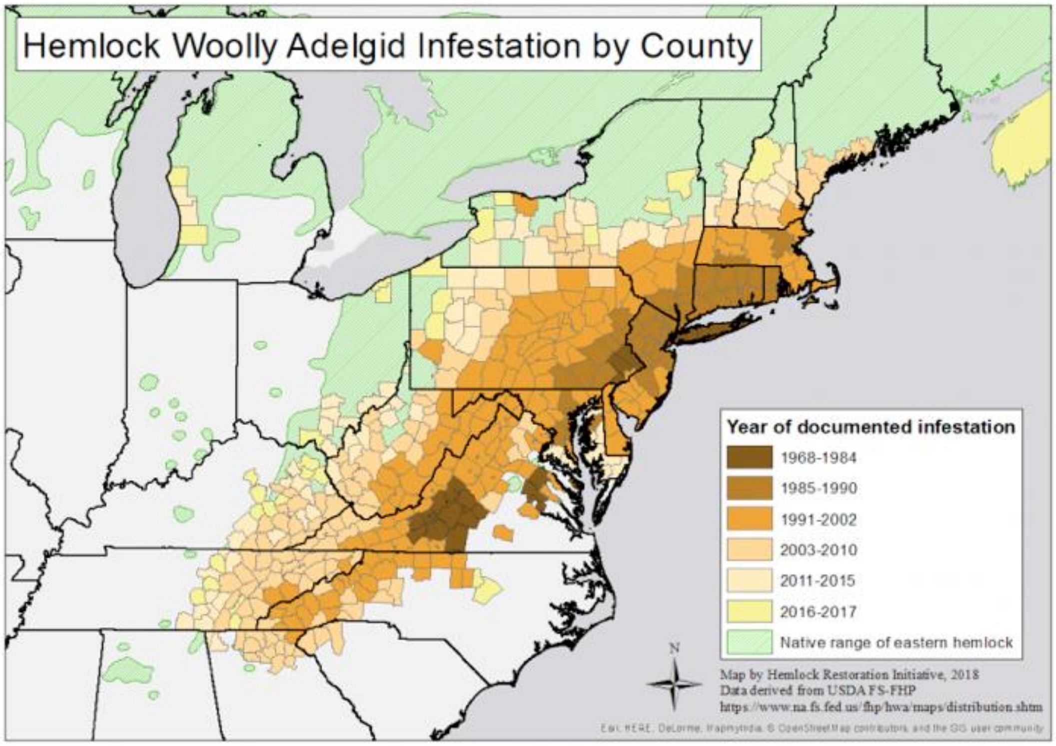

Figure 1. Hemlock Woolly Adelgid infestation by county in the United States, with year of infestation indicated by different shades of yellow.
Source(s): Factors Threatening the Appalachian Hemlock. (n.d.). Retrieved from https://savehemlocksnc.org/factors-threatening-the-appalachian-hemlock/

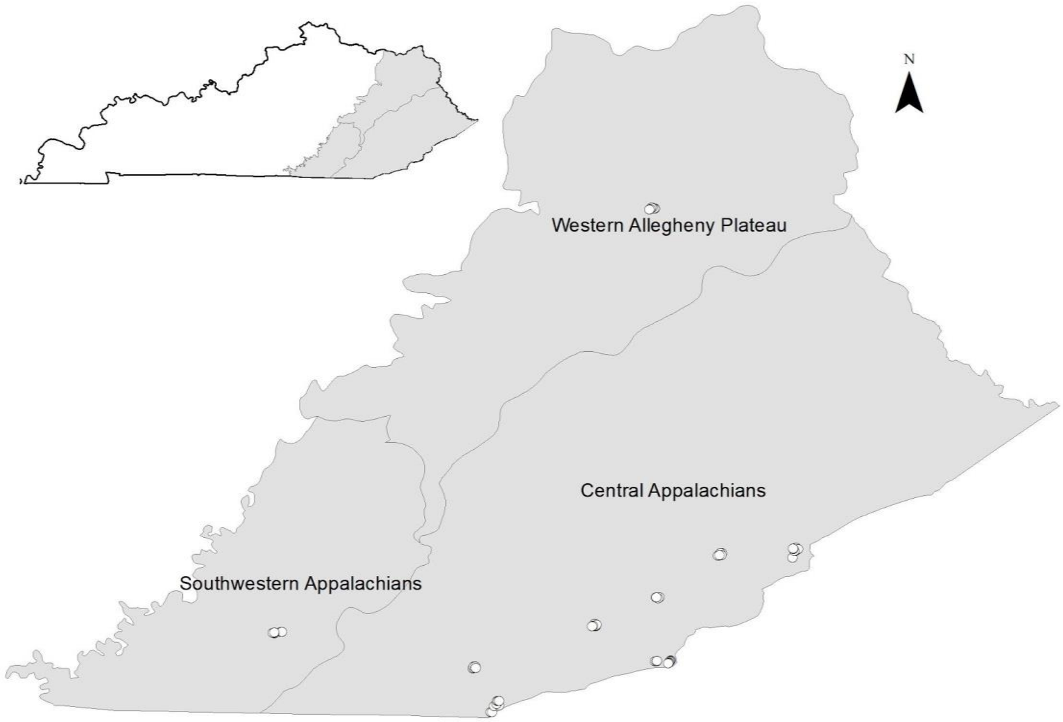

Figure 2. The Appalachian Mountains ecoregion of eastern Kentucky with 65 study sites (open circles) used in 2009 and 2018.

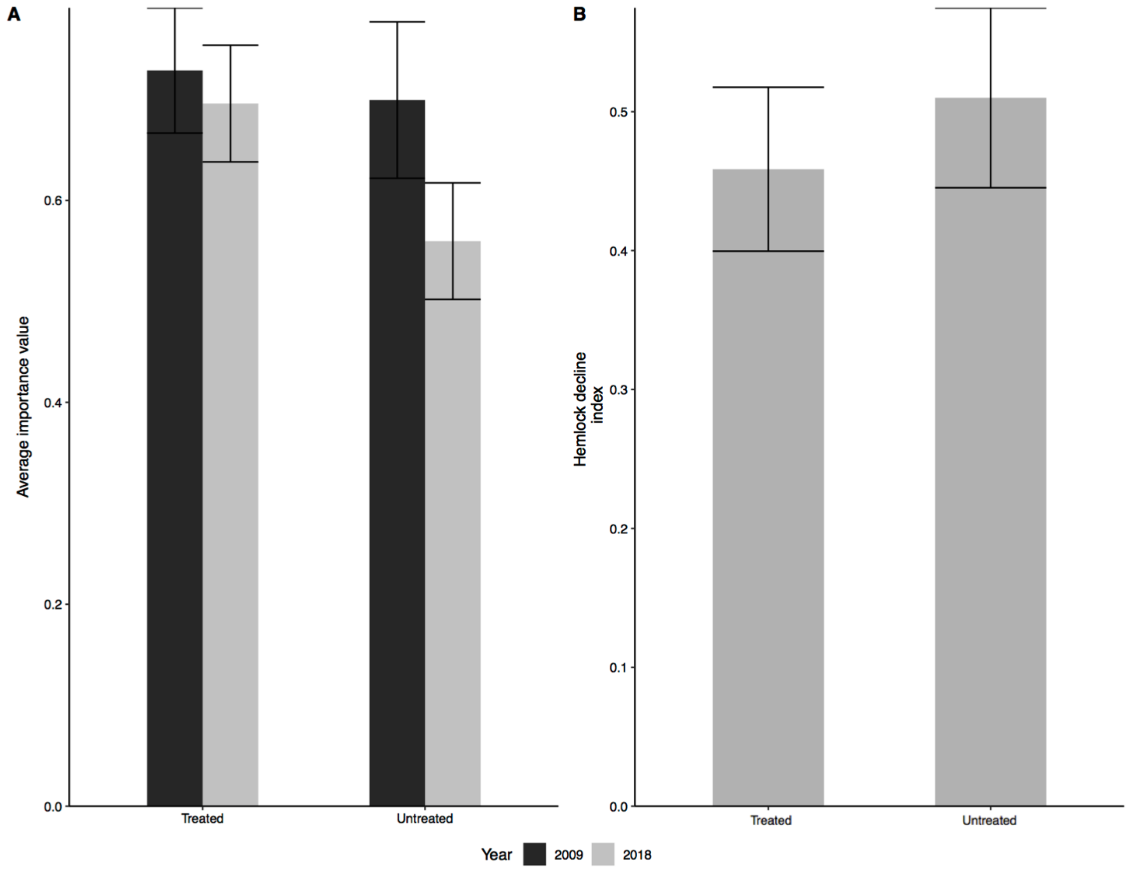

Figure 3. Mean (± SE) (A) importance value in treated and untreated sites between 2009 and 2018 and (B) hemlock decline index in treated and untreated sites.

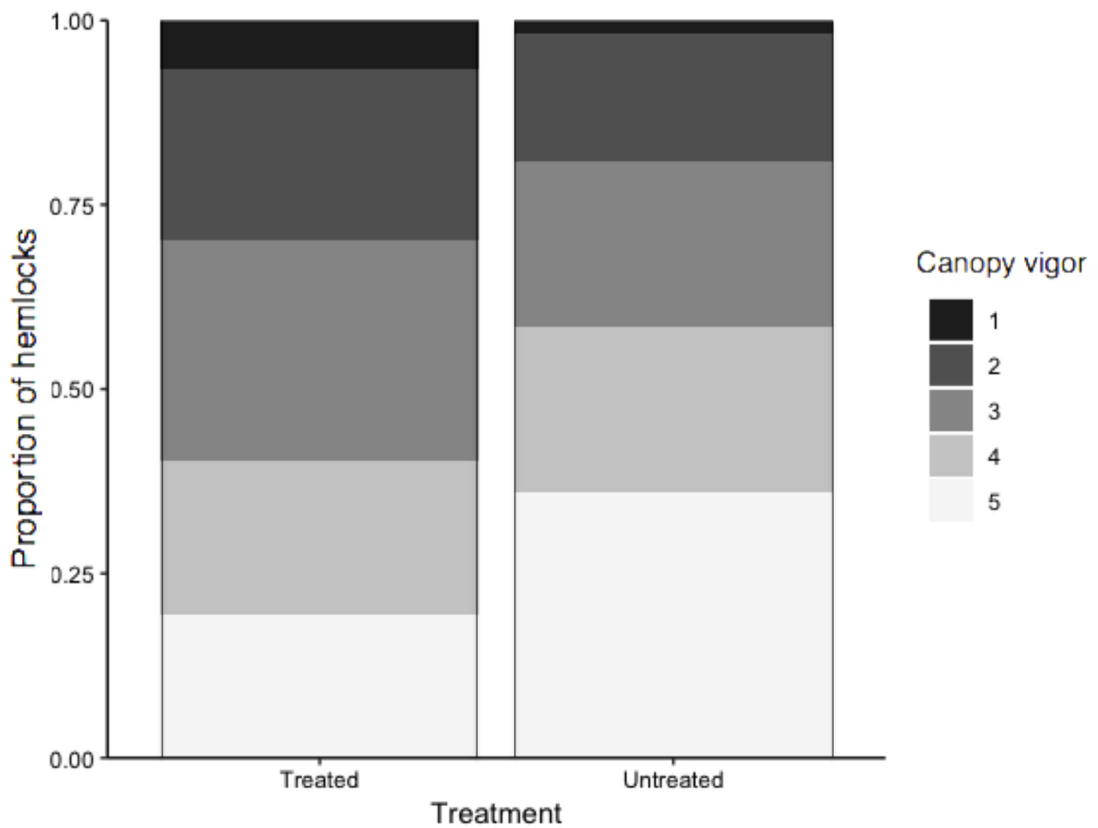

Figure 4. Proportion of Eastern hemlock trees within chemically treated and untreated sites classified with a 1-5 (1 = 76–100%, 2 = 51–75%, 3 = 26–50%, 4 = 1–25%, 5 = Dead) canopy vigor rating.

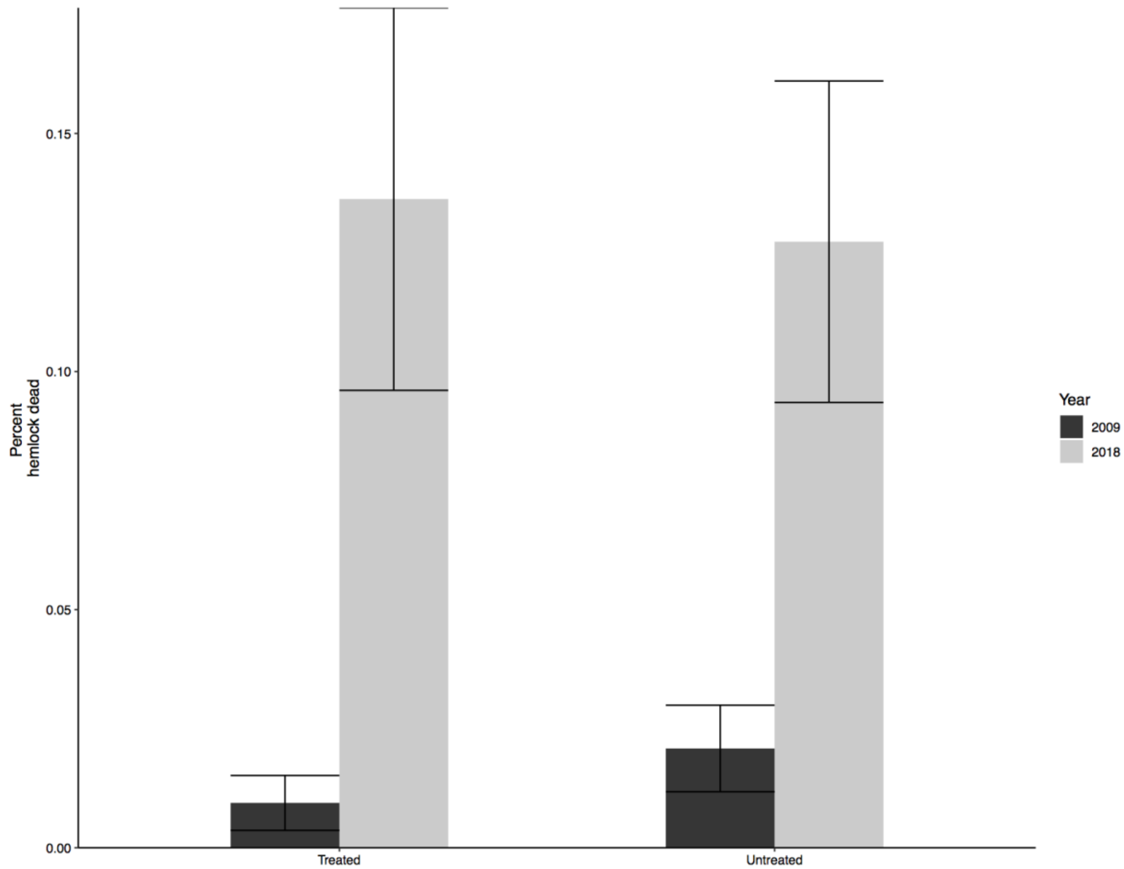

Figure 5. Percent dead hemlock between (± SE) years in chemically treated and untreated sites.

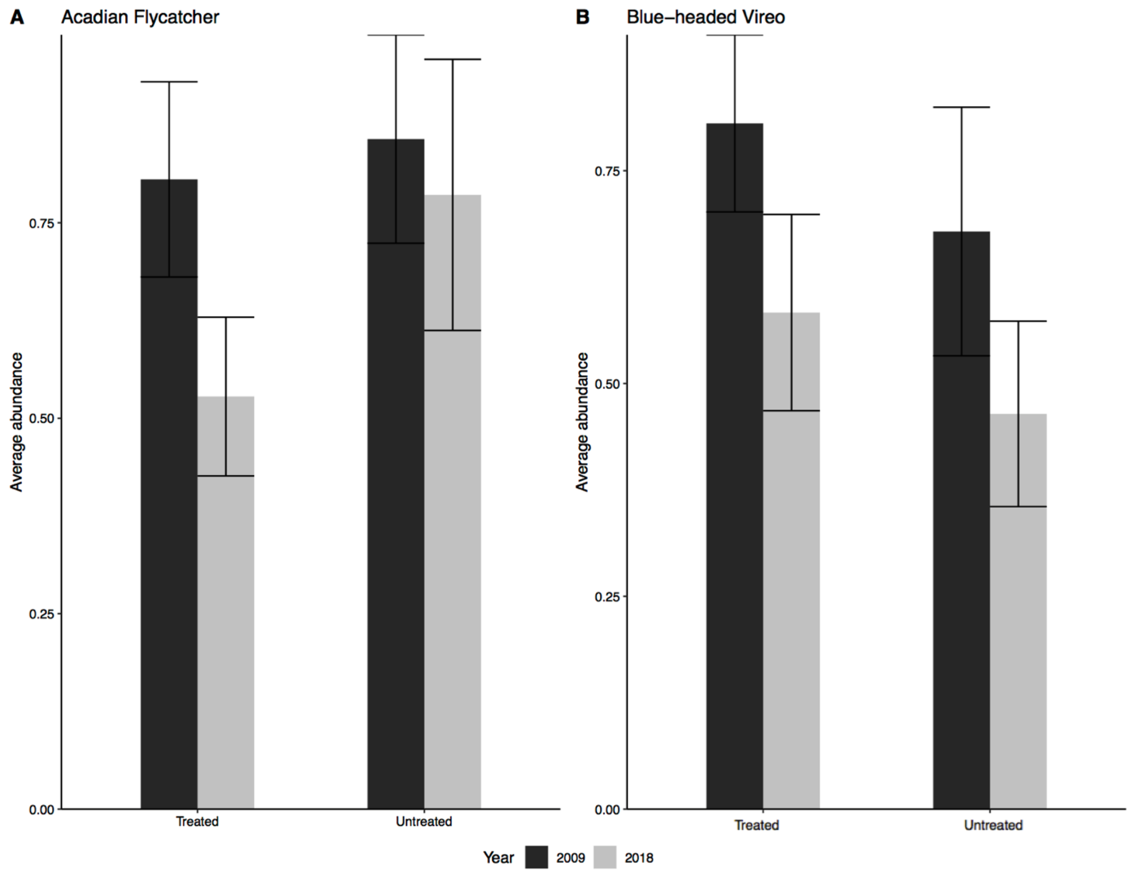

Figure 6. Mean abundance (± SE) of two species predicted to be positively associated with hemlock: (A) Acadian Flycatcher and (B) Blue-headed Vireo in treated and untreated sites in 2009 and 2018.

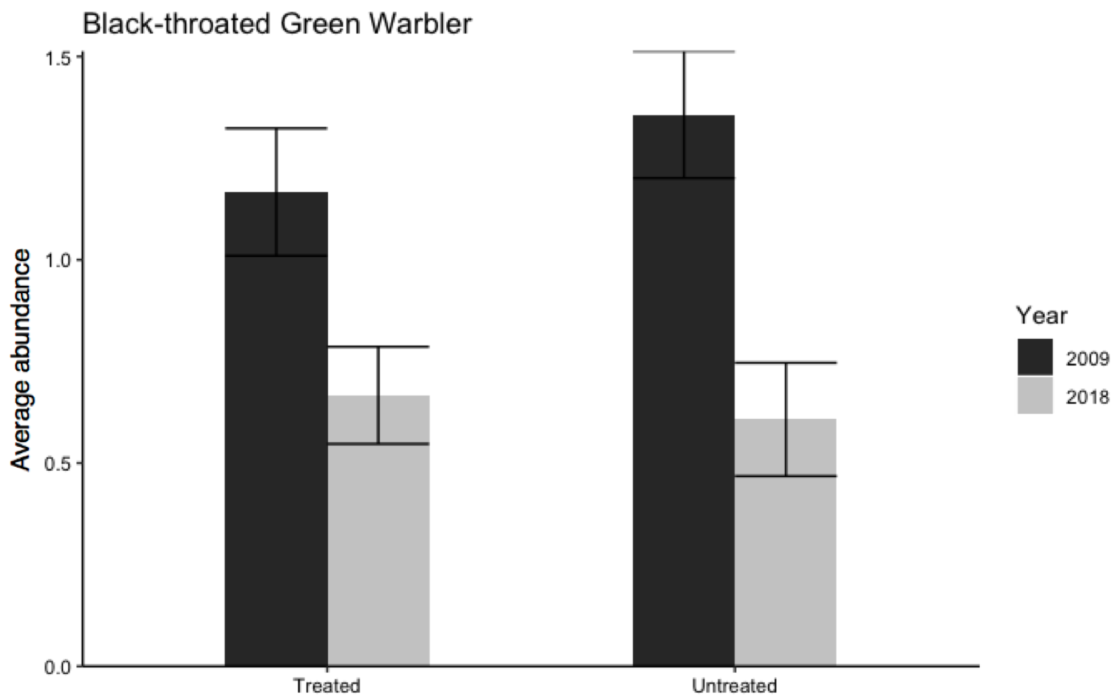

Figure 7. Mean Abundance (± SE) of Black-throated Green Warbler, a species positively associated with hemlock in 2009 and 2018 in chemically treated and untreated sites.

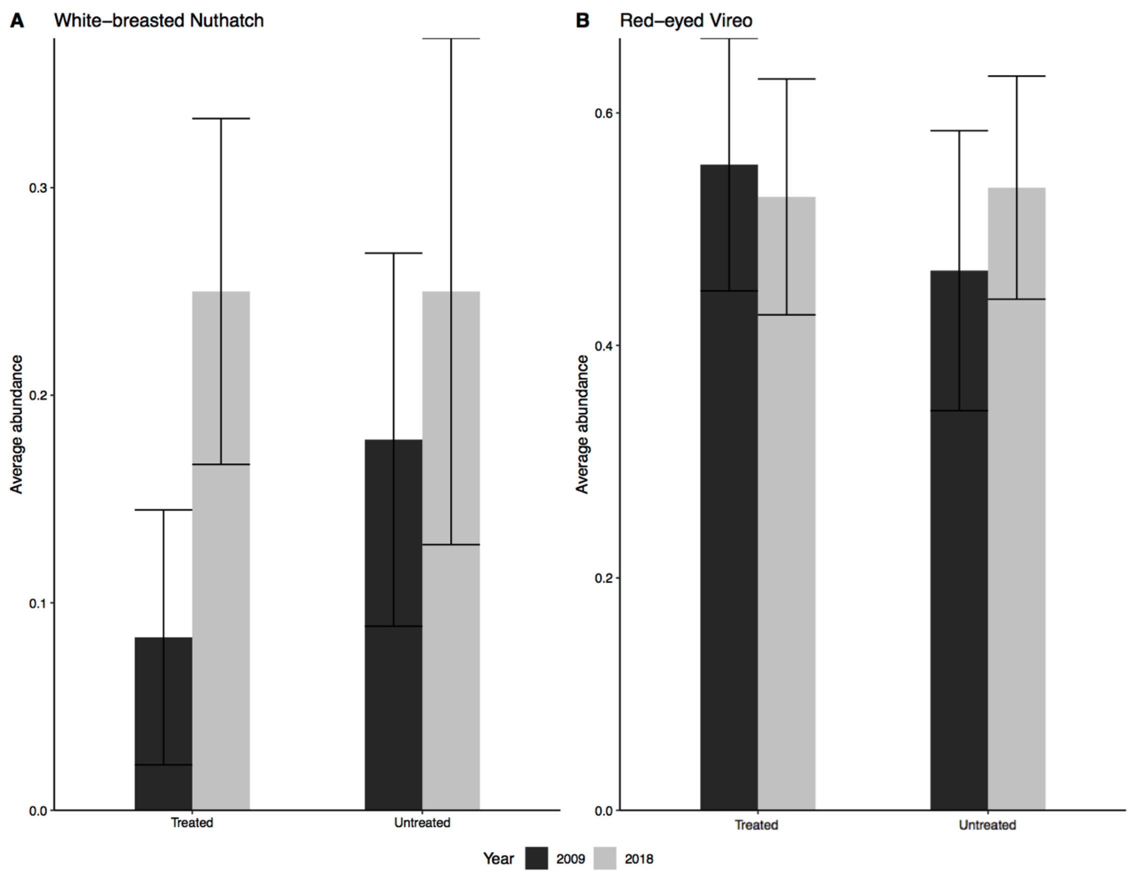

Figure 8. Mean abundance (± SE) of two species predicted to be negatively associated with hemlock: (A) Red-eyed Vireo and (B) White-breasted Nuthatch in treated and untreated sites in 2009 and 2018.

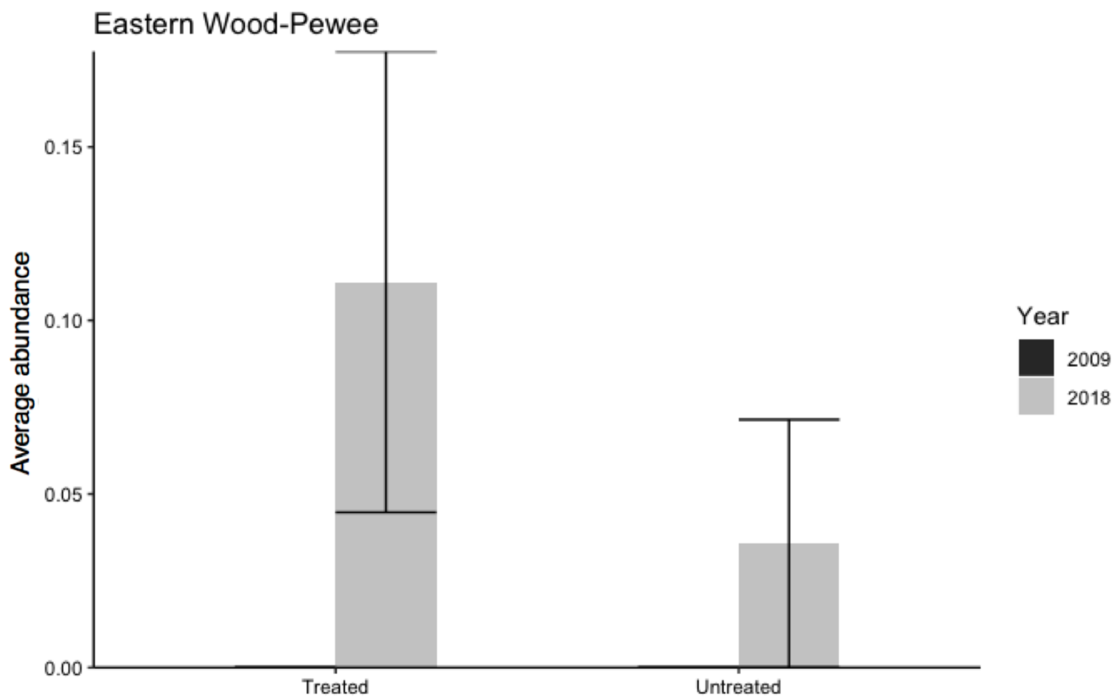

Figure 9. Mean Abundance (± SE) of Eastern Wood-Pewee in 2009 and 2018 in chemically treated and untreated sites. There were zero detections in 2009

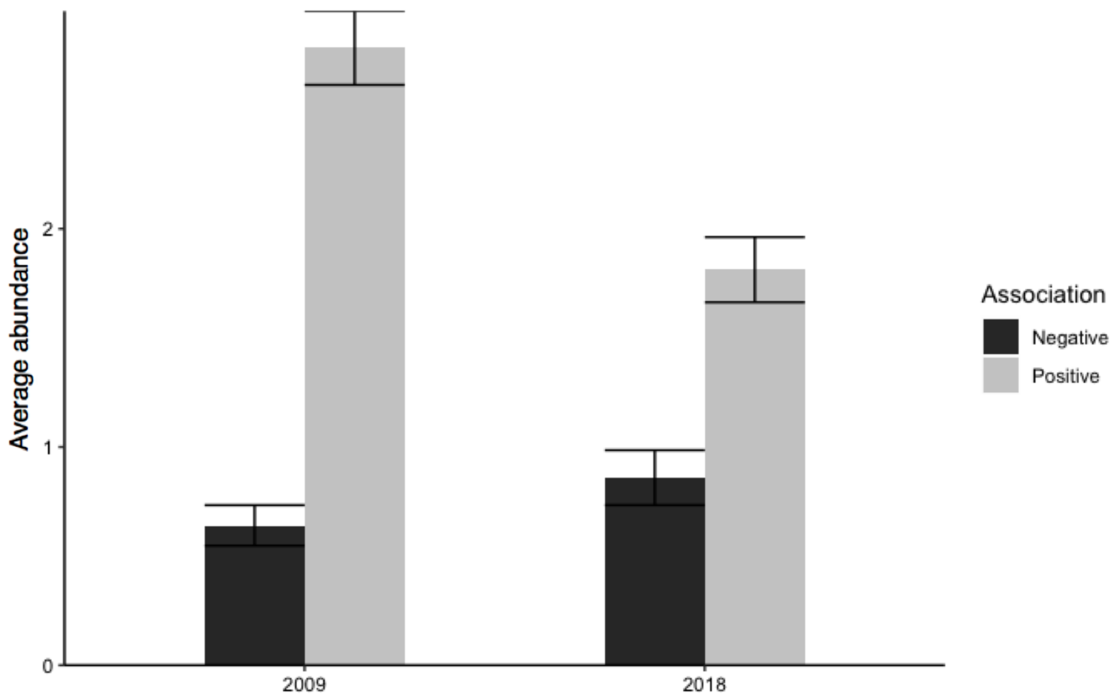

Figure 10. Average abundance (± SE) of avian species in 2009 and 2018 grouped by positive or negative association with Eastern Hemlock trees.

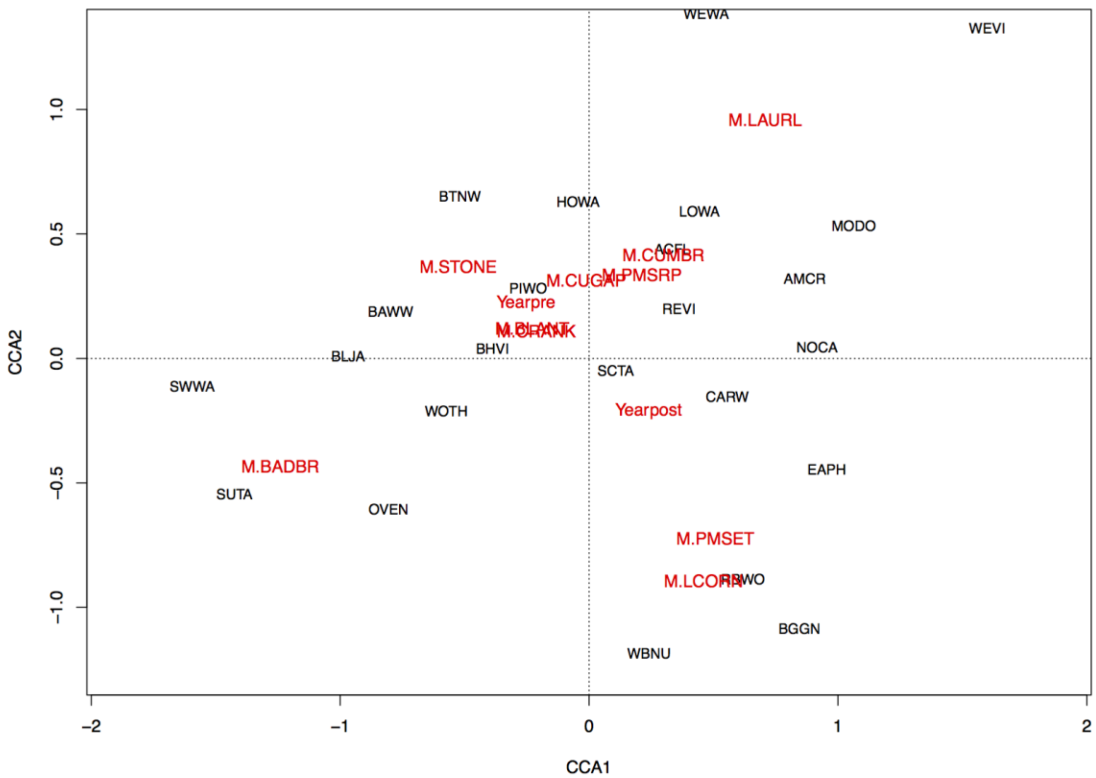

Figure 11. Canonical Correspondence Analysis (CCA) identifying correlation between avian community composition and environmental variables. M.BADBR = Bad Branch State Nature Preserve; M.BLANT = Blanton Forest State Nature Preserve; M.CRANK = Cranks Creek Wildlife Management Area; M.CUMBR = Cumberland Falls State Resort Park; M.CUGAP = Cumberland Gap National Historical Park; M.LAURL = Laurel Gorge Wildlife Management Area; M.LCORN = Lilly Cornett Woods; M.PMSET = Pine Mountain Settlement School; M.PMSRP = Pine Mountain State Resort Park; M.STONE = Stone Mountain Wildlife Management Area; Yearpre = 2009; Yearpost = 2018; ACFL = Acadian Flycatcher; AMCR = American Crow; BAWW = Black-and-white Warbler; BGGN = Blue-gray Gnatchatcher; BHVI = Blue-headed Vireo; BLJA = Blue Jay; BTNW = Black-throated Green Warbler; CARW = Carolina Wren; EAPH = Eastern Phoebe; HOWA = Hooded Warbler; LOWA = Louisiana Waterthrush; MODO = Mourning Dove; NOCA = Northern Cardinal; OVEN = Ovenbird; PIWO = Pileated Woodpecker; RBWO = Red-bellied Woodpecker; REVI = Red-eyed Vireo; SCTA = Scarlet Tanager; SUTH = Summer Tanager; SWWA = Swainson's Warbler; WBNU = White-breasted Nuthatch; WEVI = White-eyed Vireo; WEWA = Worm-eating Warbler; WOTH = Wood Thrush.