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The response of an avian community to intercropping and forest management practices in a private working pine forest

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The response of an avian community to intercropping and forest management practices in a private working pine forest

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Within managed pine forest systems, a plethora of bird species exist throughout the lifecycle of a stand akin to what may be experienced through post-disturbance regeneration in a natural forest system. I sought to address how breeding avian communities shift across time in response to stand aging and forest management, evaluate species-specific responses to stand conditions, investigate the responses of at-risk avian species to forest management, and determine avian non-breeding, over-wintering presence in a managed loblolly pine (*Pinus taeda*) forest.

I conducted breeding bird point count and vegetation surveys within five stands of privately owned and managed pine forest in Mississippi, each of which was split into quadrants with different management strategies implemented. I designed and executed night surveys for Chuck-will's-widow (*Antrostomus carolinensis*). Lastly, I conducted winter bird banding to explore over-wintering diversity, dietary isotope assimilation, and parasite prevalence.

I found evidence that the avian community shifted in response to forest stage and structure, with differences created by management practices and forest succession. Switchgrass (*Panicum virgatum*) intercropping allowed some grassland and early successional species to

remain in forest stands with closing canopies longer than in plots with standard management, with few diversity differences between treatments. Abundance of Chuck-will's-widow was found to relate positively with the percent cover of early successional forest stands, those which were recently harvested and replanted and were in a pine-grassland state. Over-wintering bird species richness remained relatively low, and capture rates were consistently greatest in a young open canopy stand, which contained a higher level of vegetation structure and plant abundance when compared to three older stands.

This represents a limited number of studies where investigations into bird community responses to forest management took place in the same forest stands across a long temporal period. Managers in forest systems should consider the implications of management undertaken at different stages in the rotational lifetime of a forest stand. To focus on conservation of priority bird species, managers should increase heterogeneity by maintaining or creating pine-grassland and early successional forest conditions within forest stands while also ensuring stands of various ages exist concurrently within the forest ecosystem.

DEDICATION

For my parents, Carla and Mike, and my grandparents, Grandma Marcy and Buddy.

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TABLE OF CONTENTS

CHAPTER

LIST OF TABLES

LIST OF FIGURES

CHAPTER I

INTRODUCTION

The southeastern United States is known for high biodiversity, but also supports one of the largest continuums of privately planted forests in the world (Brockerhoff et al. 2008, Graham et al. 2010, Jenkins et al. 2015, Evans et al. 2021). Historically, bird diversity was documented to be lower in these managed pine systems than in hardwood or pine-hardwood forests (Anderson 1975, Dickson and Segelquist 1979, Shackelford and Conner 1996), but more recent studies have demonstrated that pine dominated landscapes can support high levels of avian species richness (e.g., Loehle et al. 2005, Parrish et al. 2017). While it was suspected that high avian richness levels were a direct result of the pine-hardwood dichotomy, recent studies have shown that a return to pine-grassland and early successional pine systems, when compared to created pine-hardwood forests that had replaced the historical systems due to fire suppression, leads to an increase in overall bird diversity (Ware et al. 1993, Wilson et al. 1995, Gorden et al. 2001, Loehle et al. 2005). Increasing management activities within these working forests, whether through fire, mechanical thinning, or herbicide use, commonly leads to increasing amounts of pine forest structural heterogeneity and, consequently, positive diversity responses by the avian community (Gordon et al. 2001, Conner et al. 2002). In addition to this finding, it has been suggested that the avian species richness is directly related to the increasing amount of structural diversity and complexity as a result of constructing a forest landscape with stands of varying age classes through management actions (Loehle et al. 2005).

Forests managed for timber production are highly variable in composition and structure throughout a stand rotation, possibly causing long- and short-term effects on forest species (Chaudhary et al. 2016). Management actions can alter avian communities in multiple ways. Management goals, specifically management intensity, site preparation strategies, prescribed fire, and herbicide use, tend to determine growth of early successional plant communities in many systems (Miller and Chamberlain 2008, Jones et al. 2009). Subtle differences in forest structure and composition, created by management actions at the local and landscape levels, can cause birds to concentrate in, or avoid specific conditions.

Forestry practices in the southeastern United States are often thought to lead to a simplified forest structure through the use of clearcut harvesting (Rosenvald and Lohmus 2008, Fedrowitz et al. 2014, Chaudhary et al. 2016). Modifications such as clearcutting can affect the ecosystem at multiple scales, with bird assemblages in unaltered stands being shaped by nearby management actions (Lindenmayer et al. 2015). But small-scale management, or management at the patch or stand level, produces variations in forest vegetation configuration and composition (Linden and Roloff 2013, Kroll et al. 2014). Forest management that alters size, shape, age, and configuration of patches can affect avian species, with bird species responding to forest management actions in different ways, often from increasing edge effects (Brown and Sullivan 2005). Logging practices change environmental conditions - such as amount of light penetration and wind speed - which in turn can reduce availability of shelter and other resources used by forest bird species (Heithecker and Halpern 2007, Robertson and Hutto 2007).

Effects of clearcut harvesting and site preparation methods may be different not only for different species, but across time. Both actions can shape distinct avian communities through creation of different vegetation communities and forest conditions, though observable benefits of this process on grassland and shrubland species likely decline as clearcuts regenerate (Keller et al. 2003, George et al. 2019). With implementation of non-traditional management methods, such as intercropping switchgrass (*Panicum virgatum*) within pine rows as a biofuel crop, studies highlight the importance of expanding research temporally when investigating wildlife diversity responses to infrequent management that may not only change the vegetation structure of the stand, but water quality, soil composition, and growth of the managed pine (Homyack et al. 2013, Muwamba et al. 2015, Tian et al. 2015, Campbell et al. 2016). Many aerial insectivores have demonstrated an affinity for open forest conditions and forest edges, both of which are frequently created by active management within working pine systems (Tozer et. al 2014, English et al. 2017).

Avian species diversity and abundance are known to be greater in forests than open fields of similar size, a phenomenon explained by the increasing amount of vertical foliage complexity and foraging niches available (e.g., MacArthur and MacArthur 1961, MacArthur et al. 1962, Karr and Roth 1971, Martin and Karr 1986). Vegetation composition and structure variations cause observable differences in local avian community structure, with species associations frequently shaped by specific food resources, amount of cover, and apparent predation risk (Rotenberry 1985, Griesser and Nystrand 2009, Kroll et al. 2014). Increasing plant diversity has been correlated to higher insect diversity, which could positively affect the associated avian community in response to potential increases in food sources (Southwood 1979, Stamps and Linit 1997, Haddad et al. 2001, Wenninger and Inouye 2008, Robertson et al. 2011). Plant diversity can be an important forest characteristic when attempting to manage specifically for avian diversity (Harper 2007, Swanson et al. 2011), and can be managed in multiple ways, often through use of herbicide and prescribed fire. The heterogeneous nature of forest composition creates opportunities for foraging birds that may have specific plant species or structural requirements (Ding et al. 2008). As such, most forest management aimed at maintaining, or increasing, biodiversity focuses on conserving structural complexity within stands and providing several different seral stages within the landscape by allowing succession processes to take place (Verschuyl et al. 2008).

Sustainable forest management (e.g., Sustainable Forestry Initiative [2022] as a model) become crucial to the conservation of migratory and declining avian species (Hollie et al. 2020), but few studies have assessed the potential implications of these practices across large geographic areas and across long temporal periods (Faaborg et al. 2010, Hollie et al. 2020). Many short-term studies and meta-analyses have examined breeding bird communities in working forests, but few have attempted to quantify species diversity and forest management impacts on over-wintering species (Loehle et al. 2005, Loehle et al. 2009, Greene et al. 2016, Grodsky et al. 2016, Wood et al. 2020, Sklarczyk 2021). One such study drew attention to higher levels of richness and abundance in wintering avian species in the years following stand initiation in areas of less intensive forest management (Hanberry et al. 2013). However, within landscapes containing structurally different forest stands, winter abundances and diversity may not exhibit marked differences across time, as documented in multiple heterogeneous ecosystems (e.g., Noble and Hamilton 1976, Conner et al. 1979, Childers et al. 1986, Grodsky et al. 2016).

Assessing dissimilarities in avian community composition between different working forest ecosystems and the larger landscape can lead to discussions of effects of different management practices and forest composition and structure (Socolar et al. 2016). Species richness is a common proxy for assessing avian diversity within a specific geographic location, largely related to the ease of acquiring data (von Euler 1999). However, using species richness as the sole indicator of ecological importance of one geographic area over another and across time is generally not suitable, and studies should refrain from using richness as the only metric for evaluating effects of forest and land management strategies (von Euler 1999, Hillebrand et al. 2018). Specifically, while species richness is commonly influenced by patch size, observed values of richness do not consider which species are present (*de facto* conservation needs of those species) and their abundance (MacArthur and Wilson 1967, Fleishman et al. 2006, Banks-Leite 2012). Therefore, it was proposed by MacArthur and MacArthur (1961) that bird species diversity should replace species richness when considering what influences species presence in the environment.

While species richness may not be a true indictor of local breeding or wintering habitat quality and effectiveness of land management, community turnover rates can be suggestive of the effects of deterministic processes such as the amount of competition and environmental effects (Stegen et al. 2013). Turnover has been shown to increase as habitat heterogeneity increases (Anderson et al. 2006, Veech and Crist 2007), an effect that could be assumed to take place within large working landscapes under rotational harvest practices. Nonetheless, richness, diversity, and turnover metrics are frequently used to understanding avian community dynamics across time and space.

Objectives

I studied effects on bird communities from intercropping switchgrass and long-term active management, in both the breeding and non-breeding seasons. Although the effects of switchgrass intercropping have been examined in previous studies (e.g., Loman 2014, Marshall 2016, Fuller-Morris 2018), little research exists exploring the consequences of this technique when active management ceases. In a similar fashion, management actions such as herbicide use and prescribed fires have been heavily studied in the southeastern U.S. and in this study system (e.g., Iglay 2010), but few studies have aimed to compare the effects of multiple management actions across the lifetime of a working pine stand. Finally, few data exist examining how Chuck-will's-widow, an understudied aerial insectivore commonly documented in forested landscapes, use managed forest systems and the different forest conditions found within.

Therefore, to understand how bird communities respond to a variety of management actions throughout a full stand rotation, across time, and between seasons, I addressed the following research questions, which correspond to the chapters of this dissertation:

1. What are the effects of silvicultural methods on an avian community across a longitudinal study?
2. How is avian abundance and diversity influenced by switchgrass intercropping within an intensively managed pine forest once switchgrass management has ended?
3. Can winter avian species diversity, parasite prevalence, and dietary components be attributed to differences in forest stand structure and composition based on forest stand age?
4. Which forest stand and landscape properties, including stand age, management status, and composition, influence the abundance of Chuck-will’s-widow, a species for which little context for effective conservation currently exists?

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CHAPTER II

RESPONSE OF AVIAN COMMUNITIES TO DIFFERENT FOREST MANAGEMENT STRATEGIES ACROSS A LONGITUDINAL STUDY

Many North American avian populations have exhibited significant declines in the past 50 years (Rosenberg et al. 2019). Birds found in all forest types in the United States have decreased in abundance by 22% and grassland species decreased in abundance by 53% since 1970 (NABCI 2019). Of the species typically found in temperate forests and grasslands, 22% and 27%, respectively, are of high conservation concern, with many showing steep declines in population size (NABCI 2016). More concerning, and more recent, evidence indicates that declines are frequently observed in common and wide-spread species, not just in rare species, leading to major implications when applying conservation efforts in a variety of ecosystems (Rosenberg et al. 2019).

Loss of breeding and wintering habitat and landscape modification are two of the most significant and heavily studied factors attributed to these declines. Across much of the United States, a decrease in early successional conditions caused by land use changes and other anthropogenic factors have led to declines in populations of some avian species (King and Schlossberg 2014). In areas of frequent modification, such as within managed forests, trajectories of contained avian populations are often understudied due to the ephemeral nature of the ecological systems they inhabit (Schmiegelow and Mönkkönen 2002). In the southeastern United States specifically, changes in landcover have been identified as a primary factor in declines of the populations of some bird species; however, an increase in forest management efforts creating landscape heterogeneity and pine-grassland systems has increased benefits to some avian communities (Loehle et al. 2005).

Landscape-level characteristics are frequently documented as having a strong influence on abundance of birds within forest landscapes. However, some studies have shown that avian species richness within a landscape may not change despite modifications in understory vegetation structure and cover in managed forest stands, particularly as the total forested landscape heterogeneity increases (Dornelas et al. 2014, Lindenmayer et al. 2015), as long as the forest stands in which they are embedded provide sufficient food, cover, and substrates for daily activities. But it is important to note that forests are not static; they are dynamic in both time and space and can be affected by other organisms within the community and abiotic properties and events at that location (Gleason 1909, Kuusipalo 1985, Tonteri et al. 1990, Haila et al. 1993).

For some songbird species routinely found in managed forest stands close to harvest, landscape-level changes may be less important when compared to local stand modifications (Lichstein et al. 2002). Forest management activities frequently alter stand-level factors such as size, forest age, configuration, and plant composition and structure, which may affect avian communities at multiple scales (Herrando and Brotons 2002, Seoane et al. 2004). As forest stands are altered in size, shape, structure, and age, bird species composition can exhibit changes consistent with MacArthur and Wilson’s theory of island biogeography (1967). Decreasing stand sizes often correlate to decreasing avian taxonomic and functional diversity within the stand, with an increasing edge effect that influences avian distributions (McIntyre 1995, Hawrot and Niemi 1996, Ehlers Smith et al. 2018). Recently harvested stands, particularly those harvested the previous year and subsequently replanted, create forest- and stand-level structural changes that generate high species turnover rates and can lead to reductions in avian abundances among some species and increases among others (Dickson et al. 1993). Subsequent forest regeneration, post-harvest and replanting, can play an important role in supporting many early successional species conservation, demonstrating how rotational harvests are of particular use in evaluating localized species response to landscape changes (Bowen et al. 2009).

Within southeastern forests, managing for pine at different stages creates a rotation of vegetation conditions and cover types across time, resulting in use by early successional species at multiple time periods during the management cycle (Evans et al. 2021). This natural, or human mediated, forest succession is a temporal process that exhibits both fast and slow periods of changes in diversity and species richness and can affect avian communities at multiple scales. Silviculture practices can increase or decrease forest heterogeneity, affecting the availability and configuration of biotic and abiotic conditions within these forest systems. Forest management practices that introduce periodic disturbances (e.g., forest thinning, stand establishment) can therefore correlate with changes in avian richness across the landscape (Mitchell et al. 2006).

Twenty years ago, nearly half of the forests in the world were considered available for timber harvest, with most wood production originating from northern temperate forests (Brawn et al. 2001). In the past 30 years, the number of planted forests across the world has more than doubled, with 45% of those being plantation style forests (FAO and UNEP 2020), forests generally composed of one or two dominant tree species planted for production purposes. The southeastern United States holds the greatest extent of pine forests used for timber production within the country, with more wood products produced from this region than all the remaining regions in the United States combined (Haynes 2002). Of the harvestable forests in the southeastern United States, 19.3 million acres are contained within Mississippi (down 1.4% since 2006), with 68% (down from 72% in 2006) comprised of naturally regenerated stands (Oswalt et al. 2009, Oswalt 2017). Most managed southern loblolly (*Pinus taeda*) pine forests are comprised of stands aging from 1–20 years (Oswalt et al. 2009) and are managed on a less than 30-year rotation. However, more recent management actions have resulted in sizeable increases in large diameter, late rotation trees (21–28.9in), with volumes five times higher than in 1977 and continuing to proportionally increase (Oswalt 2019). Within these systems, young forests, classified broadly as early successional or early establishment forests, can support a high diversity of bird species as the constant rotation and creation of young stands creates a landscape mosaic favorable to their life history needs (Dickson et al. 1993). Peak avian species diversity or total avian density has been documented within 12 years of logging with the lowest levels of diversity documented shortly after a disturbance event such as harvesting (Venier and Pearce 2005, Ding et al. 2008).

Forests in the early stages of regrowth are of great ecological value, specifically as they are categorized by a high diversity of vegetation structure (Hagan and Meehan 2002, King and Schlossberg 2014). Shrub cover and the proportion of ground foliage peak early in the years of forest growth within managed forests and tend to decrease as canopy cover increases (Ding et al. 2008). Where vegetation composition and vertical structure remain, the heterogeneous nature of forest composition creates opportunities for foraging birds that may exhibit positive relationships with plant species or vegetation structure (Ding et al. 2008). Young forests containing dense understories often exhibit high avian diversity and densities, compared to mid-age stands with closing canopies that shade out lower vegetation (Dickson et al. 1993). Structural richness created by the natural or anthropogenically manipulated rotation of plant species, from annual herbaceous species to shrubs and slower growing hardwoods, allows for a broader range of species to benefit from available forest resources (Swanson et al. 2011). Positive effects of vegetation cover and basal area are documented for many forest interior and edge species within these systems (Lee and Carroll 2014).

With working forests providing many ecosystem services, including support to breeding and wintering bird populations, disturbance through management has the potential to perpetuate biodiversity (Vogt et al. 1999, White and Jentsch 2001, Iglay 2010). Site preparation intensity, mid to late rotation management, and non-traditional management methods such as intercropping, have demonstrable impacts on avian communities (e.g., Kilgo et al. 2000, Iglay 2012, Singleton et al. 2012, Loman et al. 2013, Loman et al. 2014, Greene et al. 2016, Grodksy et al. 2016, Evans et al. 2021; but see Owens et al. 2014). However, an understanding of the full cycle of a forest stand, from site preparation and planting through harvest, and its effects on avian communities are rarely studied in many forested environments.

Studies focusing on changes in species diversity among different forest stand conditions, especially as vegetation structure and management change over time, are greatly needed to aid in conservation efforts of many forest-dwelling passerines. Therefore, I aimed to quantify bird community richness, diversity, and turnover rates across a 21-year period, not only in response to forest stand stage (years post initiation grouped by similarities) but also by the specific management action used. I examined how stand structure, especially changes in vegetation composition and canopy cover, and stand stage influenced the presence of priority bird species according to the Eastern Gulf Coastal Plan Joint Venture (Greene et al. 2021). I used both breeding bird and vegetation survey results to model avian species abundance across time and yearly differences between differently managed forest stands.

It is well established that open forest conditions can support many grassland, early successional, and shrub/scrub bird species within working forests, but these species are generally less likely to be observed when the forest canopy closes unless intermediate actions are applied (see Evans et al. 2021 for discussion of intermediate management effects on early successional bird species). I anticipated grassland and early successional species would respond positively to management across all stages, specifically switchgrass intercropping, prescribed fire, and mechanical thinning, all of which could create the understory structure used for nesting and foraging. I predicted that the presence of priority bird species would be negatively correlated with forest stage and, consequently, suspected an inverse relationship between conservation value and forest stage, regardless of management action, as many priority species are associated with grassland and early successional vegetation conditions. I expected avian species diversity and richness to decrease post stand initiation in response to increasing understory density and a closing canopy and respond positively to use of prescribed fire most thinning. I expected the abundance of individual species would vary in response to the vegetation structure (e.g., early successional and grassland species would exhibit greater estimated abundances in young stands with open canopies and pockets of dense understory vegetation [i.e., shrub, forbs, and vine]). Finally, I anticipated that avian turnover rates would be high after stand initiation and remain high across the study specifically in years following management implementation, as avian and vegetation communities shift in response to time and forest management actions implemented.

Methods

Study Site

My study site consisted of intensively managed pine (*Pinus* spp.) stands in the Interior Flatwoods Area and Upper Coastal Plains Region of Kemper and Noxubee Counties, Mississippi

(Pettry 1977). The approximately 24,000-hectare landscape contained 70% managed pine, predominantly loblolly pine of various stand ages, 27% mature pine-hardwood or hardwood often along streams (streamside management zones; hereinafter SMZ), and 3% non-forested areas. This site has been owned and managed by Weyerhaeuser Company since 1962. Stand management consisted of clear-cut harvesting followed by site preparation and planting, including using a combination plow with a v-blade, bedding plow, and a subsoiler to establish pine beds. Seedlings were planted at a density of approximately 1100 trees per hectare, with a spacing of 1.5 x 6.1 m. Herbaceous weed control was applied on pine rows in bands, with stands later thinned to approximately 310 trees per hectare.

Experimental Design

For my study, I used five forest stands chosen from the larger managed landscape (Figure 2.1). When this study began, in 1999, six mid-rotation (16–19 years since establishment) loblolly pine stands were selected for inclusion in the experimental design, but only the five used throughout the full study were used for analyses. Each stand was commercially thinned 2–5 years prior to project initiation, 59–120 ha in size, and fertilized immediately post-thin and again in winter 2 years after project initiation following Weyerhaeuser NR Company protocols with diammonium phosphate. Stands were clear-cut harvested in 2009 and replanted in winter 2010–2011. At the time of replanting, in 2011, one stand was removed from the study, resulting in five stands maintained through the remainder of this study. In replanting, pine seedlings were planted 1.5 meters apart and spaced in rows 6.1 meters apart. Imazapyr and sulfometuron-methyl herbicides were applied during the first growing season post replanting, to reduce woody and herbaceous competition.

Each of the five study stands, or blocks, was initially randomly split in 1999 and again in 2011 into four separate treatments, each of 10 hectares. Treatments from 1999 to 2008 were: (1) Pine Control; (2) Herbicide; (3) Burn; and (4) Burn plus Herbicide. From 2011 to 2021, the treatments were: (1) Pine Control; (2) Switchgrass Intercropped; (3) Hardwood Control (since 2015); and (4) Switchgrass Monoculture, with treatment locations randomly chosen from within the same five stands. Surveys in the switchgrass monoculture treatment were discontinued one year after stand initiation and will not be considered in any analysis.

For extensive information regarding experimental treatments used from 1999–2008, see Iglay 2010. The six treatments used during the entire study were implemented as described below:

1. Pine Control (C; all years): managed with standard Weyerhaeuser operating procedures for site preparation, planting, fertilizing, and thinning.

2. Switchgrass Intercropped (SI; 2011–2021): following Weyerhaeuser standard planting procedures, including woody debris and stump removal, pine seedlings were planted in 2011, followed by switchgrass (Panicum virgatum) implementation in 2011 and 2012. Switchgrass treatments were established with two principles in mind: an evaluation of the utilities of switchgrass planting within the pine stands, and the effects of intercropping switchgrass on mammal and bird communities (Briones et al. 2013, Loman 2014, Marshall 2016). After seeding, an additional treatment of glyphosate was applied. Switchgrass was baled in 2013, 2014, and 2015. No further switchgrass management has been implemented since 2015.

3. Hardwood Control (HC; 2016–2021): Rows were chemically treated via aerial spraying to decrease growth of hardwood and herbaceous competition. Herbicide was applied in the fall of 2015; a combination of Arsenal (Imazapyr; used for hardwood control) and Escort (metsulfuron-methyl; used for blackberry control) was used (Fuller-Morris 2018).

4. Burn (B; 1999–2008): Prescribed burns were applied using drip torches in January 2000 and 2003 and February and March 2006 within recommended environmental conditions (Iglay 2010).

5. Herbicide (H; 1999–2008): Imazapyr was applied in fall 1999 following project initiation (Iglay 2010).

6. Burn plus Herbicide (BH; 1999–2008): Imazapyr was applied in the fall following project initiation. Prescribed burns were applied using drip torches in January 2000 and 2003 and February and March 2006 within recommended environmental conditions (Iglay 2010).

Avian Point Count Surveys, 2019–2021

Using protocols established in 2011, I conducted breeding bird surveys from mid-May through June; this "safe date" period allows for most non-breeding migrants to pass through prior to surveying while still accounting for residents that may start breeding earlier in the season. I conducted point count surveys at three pre-established locations along a diagonal transect within each treatment area (Figure 2.2). Locations were previously established in 2011 and used in subsequent studies (Loman 2014, Marshall 2016, Fuller-Morris 2018). Points were spaced 100 meters apart, and at least 50 meters from plot edges and SMZ. I surveyed each point five times per breeding season. As there were three points per treatment, and three treatments per block, I conducted 225 breeding season surveys each year. I used a standard ten-minute point count protocol for each survey following the methods of previous studies at the same site (Buckland et al. 2001, Buckland 2006, Loman 2014, Marshall 2016, Fuller-Morris 2018).

Surveys were conducted by two people, one observer and one recorder. Only observations made by the observer were recorded. The observer recorded all birds detected within each 10-minute survey period, recording species, sex if able to confirm, age (juvenile or adult), method of first detection (visual or auditory), time during the survey the individual was initially detected, radial distance in meters from the observer at time of detection, cardinal direction of detection, and flock size if applicable and detected concurrently. I modified the protocols used by Fuller-Morris (2018) to include all individuals detected during the surveys at any distance. From 2011–2015 (previously collected data; Loman 2014 and Marshall 2016) and 2020–2021 all detected individuals were recorded; from 2016–2019 (previously collected data and first year of my project; Fuller-Morris 2018), only individuals within 50 meters radially of the survey point were recorded. I estimated distance by using the grid spacing of the planted pine trees (1.5 x 6.1m) as a reference. I only recorded detections if an accurate distance estimate could be achieved; soaring birds, flyovers, and flythroughs that were not known to land within the survey area were not recorded. I used other vegetation and topography as visual aids in confirming distance to detected individuals.

I conducted the surveys between sunrise and 10:30 am CST. I did not survey during inclement weather or conditions that could affect detectability (i.e., high winds, rain, and fog). If weather conditions changed during a 10-minute survey, I made every effort to finish the survey appropriately; however, if conditions altered in a way that detectability was drastically changed, I stopped the survey, and repeated it once conditions were suitable.

Avian Point Count Surveys, 2011–2018

From 2011–2018, avian point count data were collected using two different protocols. From 2011 through 2015, data were collected in Switchgrass Intercropped and two Pine Control treatments. Loman (2014) and Marshall (2016) surveyed the same point count locations that were used in my surveys, recording every detection regardless of distance from point. In 2016 and 2017, Fuller-Morris (2018) surveyed the same points in each treatment, but only recorded detected individuals within 50m of each survey point. In 2018, data was collected following Fuller-Morris (2018). See Loman (2014), Marshall (2016), and Fuller-Morris (2018) for further information regarding survey design and protocols.

Avian Point Count Surveys, 1999–2008

In addition to the summer avian data collected since 2011, an additional 10 years of summer avian surveys were completed in the same forest stands, collected from 1999 through 2008. Treatments in each stand during these years were Burn only, Herbicide only, Burn plus Herbicide, and Pine Control (see Iglay 2010 for description of each treatment and the corresponding management actions, and Table 2.1 for an overview of stand survey activities). Surveyed points were placed in the four corners of each treatment, rather than three points on the diagonal. Points were located at least 75m from each forest edge and at least 100m from other bird survey points. Detections were recorded in distance bands of < 25, 25–50, and > 50m in 1999 and 2003–2008 and a fixed-radius circular point of < 50m from 2001–2002. Detections were recorded in time brackets of 1–3, 4–5, and 6–10 minutes. Surveys were conducted with up to six replicates per season (Iglay 2010).

Vegetation Surveys, 2019–2021

I followed a vegetation survey designed by Fuller-Morris (2018). I completed 25 vegetation surveys per treatment, centered along the same diagonal axis as the avian point count surveys. I centered five survey points around each point count survey location and created two additional centers halfway in between each top-middle and middle-bottom point count location. I then designated each vegetation survey point by randomly choosing a distance, up to 20m, and direction from each center point.

I estimated overstory canopy cover at each point using a concave spherical densiometer (Lemmon 1956; Forestry Suppliers, Inc., Spherical Crown Densiometer, Concave Model-C). I acquired a measurement of percent canopy cover for each of the four cardinal directions around the center of the random survey point. I then averaged these four measurements for one measure of canopy cover per survey point.

To determine ground cover density and growth form composition, I created a one-meter quadrat that was centered on each vegetation survey point. Due to the denseness of the understory in most of the plots, I used a one-meter rope centered on the survey point to create a one-meter diameter sampling circle. I estimated percent cover for the following: vine, forbs, graminoids (not including switchgrass), switchgrass, shrubs, hardwood trees, pine trees, dead vegetation, and bare ground. I recorded all estimates using the Daubenmire scale; this standardized the cover percentages as obtaining accurate estimates is difficult and varies by observer (Daubenmire 1959).

Vegetation Surveys, 1999–2008 and 2011–2018

For information regarding specific vegetation survey design for each summer season, see Iglay 2010, Loman 2014, Greene 2016, and Fuller-Morris 2018. Canopy cover was not collected 2011 through 2017, and ground cover categories were different for each study, with specific categories such as shrub only collected during specific years.

Statistical Analysis

Data Formatting

After combining all 21 years of bird survey data, I identified the necessity to standardize these data where possible as different survey methods and protocols were applied among years. I removed bird detections beyond 50m, radially, from each survey point as this was the maximum value recorded during 6 of the 21 years. I also used only the first three survey replicates per season to remove seasonal variation caused by differences in survey dates as surveys in the early portion of the project were often held late in the breeding season whereas surveys conducted later in the project concluded prior to the end of June. I removed documented flyovers and flythroughs from all models. I removed non-breeding species such as Cerulean Warbler (*Setophaga cerulea*) and Sedge Wren (*Cistothorus stellaris*), which were generally detected only once during the project and were likely late migrants.

From the remaining data, I determined the number of detections for each species by point, treatment, block (forest stand), and year. I determined the mean number of detections per survey point per treatment, block, and year, and grouped values into forest stages by similar vegetation conditions to visualize number of detections in response to forest succession. I classified data from 2011–2015 as Young Open Canopy (the first five years post stand establishment), years 2016–2021 as Mid Stage, and 1999–2008 as Late Stage (post thinning; see Figure 2.3 for example of conditions within each stage).

Richness, Diversity, Turnover, and Conservation Value

All analysis was completed in R version 4.0.3 (R Core Team 2020). All plots were created using the *ggplot2* package (Wickham 2016). I calculated richness and Shannon-Weaver diversity indices for each year, and for each treatment within each year (Oksanen et al. 2020). I chose Shannon-Weaver diversity over Simpson diversity indices because I desired higher value placed on rare species; Simpson diversity indices, being a measure of dominance, gives more weight to common species. For both richness and diversity metrics I used raw detection counts from all years so that even species with too few detections would qualify for diversity analysis.

I used the *turnover* function in the *codyn* package in R (Hallett et al. 2020) to calculate species turnover as the forest stands increased in age. I used this function because it calculates turnover as the proportion of species either gained or lost relative to the total number of species observed between the lower and upper time periods. I did not estimate turnover rate for 2011 as the stands were recently initiated and no surveys were completed in 2010.

For each avian species, I determined a modified Partners in Flight (PIF) rank using Nuttle's method (2003). Partners in Flight rankings provide a guide for assessing the conservation status of each avian species, where a higher value indicates a higher conservation value and greater relative risk of extinction (Panjabi et al. 2020, Partners in Flight Science Committee 2020). Rank values range from 0 to 4. A higher PIF value indicates that the species uses an ecosystem type that is declining and is in need of conservation efforts and management actions. Nuttle's method uses population trend, breeding and non-breeding distribution, threats, and abundance values for each species to create a weighted, simplified PIF rank that can be used in species and ecosystem comparisons. I used the assigned modified PIF ranks and number of individuals detected for each species to create a conservation value index (CV) for each treatment within each year. Number of detections, instead of calculated abundance, was used for each species to keep from discounting the detection of rare species with insufficient numbers for abundance estimations. Each detection value was multiplied by the species PIF rank. I then added all values together and divided by the total of number of detections for that specific treatment and year (Loman 2014, Marshall 2016, Fuller-Morris 2018). I calculated CV for treatment and survey block within each year.

Index Models

I used linear mixed effects models (LMM) using the function *lmer* in R (Douglas et al. 2015) to compare diversity, richness, and CV indices among treatments. I used treatment as a fixed effect, survey block as a random effect, and set the Pine Control treatment as the reference category to estimate treatment-level effects. I used a linear model to determine if richness and diversity were different between forest stages and turnover rate varied in response to forest age. I assessed each model for normality and homogeneity of variance by plotting the model residuals and creating a Normal Q-Q plot. All statistical analyses were considered statistically significant at $\alpha < 0.05$.

For all models, I assessed model fit and dispersion by calculating c-hat in the *AICcmodavg* package (Mazerolle 2020) and retrieved the corresponding R^2 value with *r.squaredGLMM* in the MuMIn package (Barton 2020) or from within the model summary. I assessed models for normality and homogeneity of variance by plotting the model residuals and creating a Normal Q-Q plot (R Core Team 2020).

Non-metric Multidimensional Scaling Analysis and ANOSIM

To further describe avian community changes in response to forest growth, I used the *metaMDS* function in the *vegan* package in R (Oksanen et al. 2020) to complete Nonmetric Multidimensional Scaling (NMDS) analysis to evaluate how avian communities responded to forest stand age across the 21-year period. I identified dissimilarities between forest age and grouped results by stand stage. I reduced dimensionality to the first two ordination axes and plotted model results to visually assess community level trends as forest stage shifted. I plotted NMDS by both forest stage and age. I assessed fit of environmental independent variables (forest age and stage) in the ordination model using *envfit* in the *vegan* package (Oksanen et al. 2020). Finally, I used the *anosim* function in the *vegan* package to evaluate community dissimilarities between survey years, forest stages, and treatments, with 9999 permutations and a Bray-Curtis distance metric (Oksanen et al. 2020).

Priority Bird Species

Using only priority bird species identified by the Eastern Gulf Coastal Plain Joint Venture Landbird Conservation Plan (see Table 2.2; Greene et al. 2021), I combined bird and vegetation data from all years to determine which ground cover categories affected species of conservation concern. Given the number of vegetation categories used and the differences in both categories and collection method among seasons, I combined Woody and Herbaceous Vines into the category of Vine, and Grasses and Sedges/Rushes into the category of Graminoid. I chose to only use these two categories, and categories of Switchgrass, Forbs, and Forest Stand Age as my vegetation metrics as the remaining categories, including Shrub, Hardwood, Bare Ground, Moss, and Water were not consistently collected across the 21-year period.

I used a redundancy analysis (RDA) to explore the impact of my chosen exploratory variables on my response variable (species composition) and identify the best explanatory variables from the included parameters (Kindt and Coe 2005, Oksanen et al. 2020). I used number of detections for each species as my response variable and Graminoid, Switchgrass, Forbs, Vine, and Forest age as exploratory variables. I next applied an automatic stepwise model, with the *ordiR2step* function in the vegan package (Oksanen et al. 2020), for constrained ordination using permutation tests specifically designed for RDA, using distance matrices, that maximized the r-square value. Lastly, I used an analysis of variance permutation test to assess the significance of the selected constraints by applying the *anova.cca* function in the *vegan* package (Oksanen et al. 2020) to the best fit model identified by the automatic stepwise model.

Species Guilds

I used the Breeding Bird Survey database to assign a migratory and habitat guild to each species identified during surveys (Sauer et al. 2017; see Table 2.2). I used multiple LMMs to test for differences in species richness, diversity, and CV between species guilds that share traits, specifically migratory status (resident, short-distance migrant, neotropical migrant), associated breeding habitat (early successional, grassland, urban, and woodland), and species of conservation concern between treatments. Grouping species by common traits allowed me to model taxonomic group responses versus the response of individual species; the response of individual species likely varies year to year while the overall community response may remain the same during certain forest succession stages. I used treatment as a fixed effect, survey block as a random effect, and set the Pine Control treatment as the reference category to estimate treatment-level effects. For all models, I assessed model fit and dispersion by calculating c-hat in the *AICcmodavg* package (Mazerolle 2020) and retrieved the corresponding R^2 value with *r.squaredGLMM* in the MuMIn package (Barton 2020). Similarly, I used linear models to evaluate if forest stage created differences in species richness and diversity with each guild, with the Late Stage set as the reference category. I assessed models for normality and homogeneity of variance by plotting the model residuals and creating a Normal Q-Q plot (R Core Team 2020).

Species Abundance Models

Using data provided in Iglay (2010), Loman (2014), Marshall (2015), Greene (2016), and Fuller-Morris (2018), I combined 21 years of bird point count and vegetation survey data to model effects of active management at different forest stand stages. I only used detections up to 50 m to standardize effort based on detection distance across all years, and only the first three survey replicates each season. I fit the open N-mixture model of Dail and Madsen using the *pcountOpen* function in *unmarked* (Fiske and Chandler 2011) to estimate detection probability, abundance, recruitment rates, and apparent survival probability for four species. Each model equation included four components: lambda (λ), formula for initial abundance, gamma (γ), formula for recruitment rate, omega (ω), formula for apparent survival probability, and p, formula for detection probability. I modeled two woodland species and two early successional/scrub species: Common Yellowthroat (*Geothlypis trichas*), Kentucky Warbler, White-eyed Vireo (*Vireo griseus*) and Wood Thrush. Given the change in survey point locations between portions of data collection and the discontinuous nature of the management cycle (with the harvest and subsequent stand establishment in the middle of the survey years), each species was modeled twice: once with data from 2011 to 2021 and once including data from 1999 to 2008.

My response variable was number of individuals detected per survey point, by treatment and year, for each species. I used observer, temperature (°C), wind speed (km/hr), cloud cover (percent), and Julian date as detection covariates and site covariates of Shrub, Forbs, Graminoid, Switchgrass, Vine, Allgrass (graminoid and switchgrass values combined) and Canopy Cover (all in percent) as initial abundance, recruitment, and survival covariates. I tested both Poisson and zero-inflated Poisson (ZIP) mixture models to identify if the number of zeros in the data warranted use of the ZIP model. I set the model dynamic to constant. I scaled temperature and wind speed in all models. I first tested detection effects by creating single detection covariate models, and then tested all possible combinations of site covariates in combination with the chosen detection covariates. I retained models within an AIC of 2 or less of the top model. I created a global model with all covariates that had or were close to statistical significance. I assessed possible overdispersion of parameters for the global model by using *Nmix.gof.test* with 1000 bootstrap iterations (MacKenzie and Bailey 2004) to compute a Chi-square goodness of fit. If overdispersion was not identified in the global model (c-hat < 2), I ranked models by AICc in a separate model selection table (Tables 2.3 and 2.4). I included no covariate (null) and detection parameters only (if at least one covariate had a significant influence on detection) models in all model selection tables.

I retrieved yearly and treatment-based abundance estimates using *ranef*; by summing all columns, I was able to estimate posterior distributions of the random variables by year and treatment, resulting in estimates of latent abundance (Fiske and Chandler 2011). I used *confint* to extract 95% confidence intervals from included covariates (R Core Team 2020) and used the projected model abundance estimates to evaluate yearly abundance by treatment and across all sites (Fiske and Chandler 2011).

Results

Richness, Diversity, Turnover, and Conservation Value

Seventy-five species were detected between May and June during the 21-year period (Table 2.2). Seventy-three species were recorded post stand initiation prior to mid rotation thinning, and 56 were recorded post thinning prior to harvest. Of these species, 21 were of great conservation concern (modified PIF rank of ≥ 3) and 18 were considered priority bird species for the region (Greene et al. 2021). The most commonly detected species, in order, were Yellow-breasted Chat (*Icteria virens*), Indigo Bunting (*Passerina cyanea*), Eastern Towhee (*Pipilo erythrophthalmus*), White-eyed Vireo, Carolina Wren (*Thryothorus ludovicianus*), Northern Cardinal (*Cardinalis cardinalis*), Pine Warbler (*Setophaga pinus*), Common Yellowthroat (*Geothlypis trichas*), Prairie Warbler (*Setophaga discolor*), and Carolina Chickadee (*Poecile carolinensis*). The mean number of detections per survey point trended downward as forest stage increased (Figure 2.4). Mean species richness (Table 2.5) was 31.2 in the Young Open Canopy (range = 28–36, Std. dev. = 3.3), 41.8 in the Mid Stage (range = 37–50, Std. dev. = 4.8), and 41.6

in the Late Stage (range = 34–49, Std. dev. = 4.0). Mean diversity, including treatment effects, was 2.4, 2.8, and 2.8 respectively (Table 2.6).

Index Models

All LMM are presented in the corresponding tables, including estimates, confidence intervals, additional test statistics and significantly different groups. All statistics are reported as comparisons to Pine Control treatments, which follow standard Weyerhaeuser management and were not experimentally managed.

LMM results supported the significance of treatment on richness and diversity values (Table 2.7). The Burn ($P < 0.001$, $\beta = 3.67$, SE = 0.74), Burn plus Herbicide ($P < 0.001$, $\beta = 4.01$, SE = 0.74), and Herbicide ($P < 0.001$, $\beta = 2.47$, SE = 0.74) treatments had higher richness levels than Pine Control. The same treatments (Burn: $P < 0.001$, $\beta = 0.30$, SE = 0.06; Burn plus Herbicide: $P < 0.001$, $\beta = 0.21$, SE = 0.06; Herbicide: $P < 0.001$, $\beta = 0.22$, SE = 0.06) produced higher diversity index values than Pine Control values. Intercropping switchgrass had a negative influence of diversity estimates when compared to the Pine Control ($P = 0.01$, $\beta = -0.15$, SE = 0.06). Hardwood Control produced no effect on richness and diversity when compared to standard management, but negatively influenced CV ($P < 0.001$, $\beta = -0.18$, SE = 0.06). The Burn plus Herbicide treatment produced high CV estimates ($P < 0.001$, $\beta = 0.16$, SE = 0.05).

However, forest stage, independent of treatment, produced significant differences in richness ($P < 0.001$, $\beta = -8.93$, SE = 0.44) and diversity ($P < 0.001$, $\beta = -0.70$, SE = 0.04) between the Young Open Canopy and Late Stages, with lower richness and diversity values in the Young Open Canopy stage (Table 2.8). The Mid Stage also produced lower species richness when compared with the Late Stage ($P = 0.05$, $\beta = -0.83$, SE = 0.42). Both richness and diversity increased curvilinearly in response to forest age and treatment (Figure 2.5). Diversity indices, when grouped by stage, increased from the Young Open Canopy to the Late Stage (Figure 2.6).

CV estimates were significantly influenced by two treatments in LMM (Table 2.7). The Burn plus Herbicide treatment increased the CV when compared to Pine Control ($P < 0.001$, $\beta = 0.16$, SE = 0.05), while the Hardwood Control produced lower estimates ($P < 0.001$, $\beta = -0.18$, SE = 0.06). When tested independent of treatment (Table 2.8), forest stage alone revealed effects on CV, with significantly higher values in the Young Open Canopy ($P < 0.001$, $\beta = -0.20$, SE = 0.03) and significantly lower values in the Mid Stage ($P < 0.001$, $\beta = 0.30$, SE = 0.04). There was a clear decrease in CV as stands approached mid rotation, followed by an increase in value as stands were actively managed in the later mid rotation years (Figure 2.7). The highest CV estimates were consistently observed in the burn plus herbicide treatment and in the early stages of forest succession in all treatments.

Turnover rates ranged from 0.09 to 0.38, with a mean of 0.27 (Table 2.9). Turnover was affected by age, with turnover rate decreasing as the forest stands aged indicating less community structure (species and number of detections of each species) change between years ($P < 0.05$, df = 19, Std. error = 0.01; Figure 2.8). Turnover rates were also statistically different between the three forest stages ($P < 0.05$, $F_{2,18} = 3.08$), with different rates between the Young Open Canopy and Late Stage groups ($P_{adj} = 0.05$).

Non-metric Multidimensional Scaling Analysis and ANOSIM

Results of the stage based NMDS aligned with results indicated by the richness, diversity, and CV models (Figure 2.9). Using two dimensions yielded a low stress value (0.12). Using the reduced dimensional space, the permutation test evaluating the forest stage yielded an $R^2 = 0.57$ ($P < 0.01$). Assessing forest age instead of grouped stage resulted in a higher model fit (R^2 =

0.70, $P < 0.01$; Figure 2.10). Many forest-dwelling species were closely grouped in both graphs, scoring similarly on both axes, indicating similar responses to forest succession and impacts of forest management. Species commonly found in the Young Open Canopy stage exhibited a wide range of values along both axes, aligning with a larger ellipse created by the variation in the observed avian community in that stage. However, the large ellipse produced by the variation in the Young Open Canopy stage and was likely heavily influenced by data collected during the breeding season following stand initiation, as shown by the clear separation and non-overlapping ellipse in the ordination graph. As previously documented (see Loman 2014), the breeding season following stand initiation had lower species richness and species detection rates compared to next consecutive seasons, creating a dissimilar avian community.

Analysis of similarities (ANOSIM) results supported the documented differences between survey years and forest stages. Using survey year as the community, models produced a high ANOSIM statistic R value (0.76) indicating a high amount of community structure dissimilarity between years, with a corresponding significance value ($P < 0.01$). Similarly, when using forest stage as the community grouping, models found a high ANOSIM statistic R value (0.64), with $P < 0.01$. Using treatment as the community group, I found a significant difference between avian communities within treatments ($P < 0.01$), but the corresponding ANOSIM statistic R value was low (0.16), indicating fewer differences across time between treatments than between survey years and forest stages.

Priority Bird Species

The RDA focusing on only priority bird species and incorporating specific vegetation and forest age resulted in only 24.16% of the variance in observed community structure being produced by the applied metrics (Figure 2.11). Stepwise models for ordination methods indicated that all vegetation and forest age should be used when considering the variables that created the observed community structure, with an adjusted $R^2 = 0.23$. Permutation tests revealed that all vegetation metrics, specifically percent graminoid, vine, forb, and switchgrass, and forest age were significant ($P < 0.01$ for all). Forest age, as a proxy for pine growth, was the strongest predictor of priority species, as indicated by the longest arrow in the ordination graph. Directionality of forest age was also correlated with woodland priority bird species including Eastern Wood-Pewee, Kentucky Warbler, Wood Thrush, Worm-eating Warbler, and Yellow-billed Cuckoo, and one scrub species, Eastern Towhee, indicating that over half of the priority species in this system had a comparatively neutral or negative response to increasing forest succession. Prairie Warbler perfectly aligned with Graminoid in this analysis, possibly indicating an affinity for vegetation structure created by grasses, sedges, and rushes. All modeled vegetation metrics were negatively correlated with increasing forest age.

Species Guilds

For associated habitat guilds, richness was statistically significant using LMM between treatments (Table 2.10). Specifically, Burn plus Herbicide ($P < 0.001$, $\beta = 1.10$, SE = 0.36), Hardwood Control ($P < 0.001$, $\beta = 2.10$, SE = 0.41), and Switchgrass Intercropped ($P < 0.001$, $\beta = 0.76$, SE = 0.35) treatments were significant for the early successional/scrub guild, with higher richness values in the Burn plus Herbicide and Hardwood Control and lower values in the Switchgrass Intercropped treatment than in the Pine Control. Grassland and urban guilds had lower richness values in the Burn (Grassland: $P < 0.001$, $\beta = -0.40$, SE = 0.11; Urban: $P = 0.03$, $\beta = -0.25$, SE = 0.12) and Herbicide (Grassland: $P < 0.001$, $\beta = -0.44$, SE = 0.11; Urban: $P = 0.01$, $\beta = -0.31$, SE = 0.12) treatments compared to the Pine Control. Woodland species had higher richness values in the Burn ($P < 0.001$, $\beta = 3.70$, SE = 0.57), Burn plus Herbicide ($P <$

0.001, β = 3.22, SE = 0.57), and Herbicide (*P* < 0.001, β = 2.78, SE = 0.57), and lower values in the Switchgrass Intercropped (*P* < 0.001, β = -2.47, SE = 0.55). Richness was significantly higher in the Mid Stage than the Late Stage for all guilds except the woodland guild, but lower in the Young Open Canopy stage for early successional/scrub and woodland species (Table 2.11).

The LMM evaluating diversity between treatments showed similar results to the richness models, with a few differences within guilds (Table 2.12). There were higher diversity indices in all treatments except for Switchgrass Intercropped when compared to the Pine Control for the early successional/scrub guild (Burn: *P* < 0.001, β = 0.19, SE = 0.06; Burn plus Herbicide: *P* = 0.02, β = 0.14, SE = 0.0.06; Hardwood Control: *P* < 0.001, β = 0.32, SE = 0.07; Herbicide: *P* = 0.01, β = 0.16, SE = 0.06). For both grassland and urban guilds, there was a decrease in diversity in the Herbicide treatment (Grassland: *P* = 0.02, β = -0.08, SE = 0.03; Urban: *P* < 0.001, β = -0.10, SE = 0.04). For woodland species, the Switchgrass Intercropped treatment produced lower diversity values than the Pine Control (*P* < 0.001, β = -0.39, SE = 0.09), with the Burn (*P* < 0.001, β = 0.54, SE = 0.0.10), Burn plus Herbicide (*P* < 0.001, β = 0.44, SE = 0.10), and Herbicide (*P* < 0.001, β = 0.45, SE = 0.10) treatments produced higher values. Diversity indices were significantly higher in the Mid Stage, when compared to the Late Stage, for all guilds except woodland, which had lower index values, and grassland, which had no difference (Table 2.13). The Young Open Canopy created lower diversity indices than the Late Stage for early successional/scrub and woodland species, and higher values for grassland and urban species.

Richness predicted through LMMs showed statistical significance for treatment for multiple migratory guilds (Table 2.14). For resident species, the Burn (*P* < 0.001, β = 1.80, SE = 0.33), Burn plus Herbicide (*P* < 0.001, β = 2.60, SE = 0.33), and Herbicide (*P* < 0.001, β = 1.98, SE = 0.33) treatments produced higher richness numbers than the Pine Control, while

Switchgrass Intercropped ($P = 0.01$, $\beta = -0.81$ SE = 0.32) produced lower values. For neotropical migrants, the Burn ($P < 0.001$, $\beta = 2.06$, SE = 0.46), Burn plus Herbicide ($P = 0.01$, $\beta = 1.24$, SE = 0.46), and Hardwood Control ($P = 0.03$, $\beta = 1.13$, SE = 0.52) produced higher richness values than the Pine Control. There was no difference in species richness between the treatments and the Pine Control for short distance migrants. When evaluating the differences in richness between forest stages, there were lower richness values in the Mid Stage and Young Open Canopy, compared to the Late Stage, in resident species (Table 2.15). Neotropical migrants had lower richness values in the Young Open Canopy, while short distance migrants had more species present in the Mid Stage than the Late Stage.

LMM for migratory guild diversity indicated that the Burn treatment was important for all guilds, with additional treatments varying in degrees of significance (Table 2.16). For resident birds, the Burn ($P < 0.001$, $\beta = 0.34$, SE = 0.08), Burn plus Herbicide ($P < 0.001$, $\beta = 0.40$, SE = 0.08), and Herbicide ($P < 0.001$, $\beta = 0.39$, SE = 0.08) treatments had higher diversity indices than the Pine Control, while the Switchgrass Intercropped ($P = 0.01$, $\beta = -0.20$, SE = 0.07) treatment produced lower values. Neotropical migrant diversity was positively influenced by the Burn ($P < 0.001$, $\beta = 0.27$, SE = 0.06) treatment and negatively influenced by the Switchgrass Intercropped treatment ($P = 0.04$, $\beta = -0.13$, SE = 0.06). Contrastingly, short distance migrant diversity was lower in the Burn ($P = 0.04$, $\beta = -0.15$, SE = 0.07) treatment than in the Pine Control, as well as lower in the Herbicide ($P = 0.01$, $\beta = -0.19$, SE = 0.07) treatment. Diversity was lower in the Mid Stage and Young Open Canopy for the resident species and lower in the Young Open Canopy for neotropical migrants, but higher in the Mid Stage for short distance migrants, when compared to the Late Stage (Table 2.17).

For species of conservation concern, as designated by Greene et al. (2021), LMMs predicting richness and diversity showed statistically significant differences in treatment and forest stage effects (Table 2.18). Specifically, richness was significantly higher in the Burn ($P < 0.001$, $\beta = 0.86$, SE = 0.25) and Burn plus Herbicide ($P = 0.02$, $\beta = 0.56$, SE = 0.25) treatments, and diversity values were higher in the Burn ($P = 0.01$, $\beta = 0.14$, SE = 0.05) treatment. Both richness ($P < 0.001$, $\beta = -1.12$, SE = 0.20) and diversity ($P < 0.001$, $\beta = -0.21$, SE = 0.04) values were lower in the Young Open Canopy when compared to the Late Stage. However, model fit for both richness and diversity indices were poor, with $R^2 \sim 0.1$, and as such models should not be considered as explanatory.

Species Abundance Models

Common Yellowthroat

Abundance estimates ranged from 11 to 307 individuals (1999–2008 mean = 80, 2011–2021 mean = 199; Figure 2.12). Within the surveyed area, these estimates equate to density estimates ranging from 0.32–8.69 males/hectare. Estimates were consistently high from 2016 through 2021 in the Hardwood Control plots but peaked in all treatments 7 years post stand initiation (Figure 2.13a).

Post stand thinning (1999 – 2008), Common Yellowthroat responded to changes in canopy, forbs, shrub, and vine cover (Table 2.19). Increasing canopy ($P < 0.001$, $\beta = -0.085$, SE = 0.015), shrub ($P < 0.05$, $\beta = -0.034$, SE = 0.017), and vine ($P < 0.05$, $\beta = -0.022$, SE = 0.009) coverage had a significant negative influence on recruitment (Figure 2.14 a, b, and c). Increasing forb cover resulted in an increasing apparent survival probability ($P < 0.05$, $\beta = 0.045$, SE = 0.023; Figure 2.14d). Shrub was also included in the omega (apparent survival) formula but had no effect ($P = 0.09$, β =-0.059, SE = 0.034). Julian date was included in all competing models but did not influence detection probability ($P = 0.21$, β = -0.072, SE = 0.058).

Post stand initiation (2011 – 2021), increasing percent cover of allgrass ($P < 0.001$, β = 0.016, SE = 0.003) significantly increased recruitment rates (Figure 2.15a), while percent vine cover had a significant inverse relationship with apparent survival probability ($P < 0.05$, β = -0.054, SE = 0.026; Table 2.20 and Figure 2.15b). Vine was also included in the top model for recruitment, but produced no effect ($P = 0.46$, β = 0.005, SE = 0.007). Increasing Julian date ($P = 0.00$, β = 0.490, SE = 0.045) and temperature ($P < 0.01$, β = 0.036, SE = 0.014) had a positive influence on detection probability (Figure 2.15c), and though cloud was included in the top model, it did not affect the detection probability ($P = 0.16$, β = 0.001, SE = 0.001).

Kentucky Warbler

Abundance for Kentucky Warbler ranged from 0 to 270 individuals across all sites (1999–2008 mean = 182, 2011–2021 mean = 12; Figure 2.12). Density estimates range from 0–13.55 males/hectare. Abundance was consistently high in the burn treatment, increasing post thinning, and was negligible for eight years post stand initiation (Figure 2.13b).

The top model for Kentucky Warbler, post stand thinning (1999 – 2008), included graminoid and shrub covariates for recruitment (gamma) (Table 2.19). Increasing percent graminoid cover had significant inverse relationship with recruitment ($P < 0.001$, β = -0.080, SE = 0.015; Figure 2.16a), with increasing percent shrub ($P < 0.001$, β = 0.039, SE = 0.008) cover resulting in a significant increase in recruitment rate (Figure 2.16b). Probability of detection increased as Julian date increased ($P < 0.001$, β = 0.187, SE = 0.045; Figure 2.16c).

Abundance post stand initiation (2011–2021) was influenced by percent cover of allgrass and shrub, with detection influenced by cloud cover and Julian date (Table 2.20). The effects of allgrass and shrub were opposite, with increasing shrub cover significantly increasing ($P < 0.05$, $\beta = 0.030$, SE = 0.015) and allgrass cover decreasing ($P < 0.05$, $\beta = -0.024$, SE = 0.010) the recruitment rate (Figures 2.17a and b). Increasing cloud cover significantly decreased the probability of detection ($P < 0.05$, $\beta = -0.008$, SE = 0.003; Figure 2.17c), while increasing Julian date significantly increased detection probability ($P < 0.01$, $\beta = 0.367$, SE = 0.137; Figure 2.17d).

White-eyed Vireo

White-eyed Vireo abundance ranged from 0 to 327 individuals (Figure 2.12), with density estimates ranging from 0–9.26 males/hectare. Mean abundance was 158 individuals post stand thinning (1999–2008) and 202 post stand initiation. The highest estimated abundances were in stands 4 to 11 years post stand initiation. Abundance estimates were significantly higher in the switchgrass intercropped plots until the final two years of surveys (Figure 2.13c). Abundance post stand thinning decreased over time.

Post stand thinning (1999–2008), White-eyed Vireo abundance models included vine, shrub, and Julian date effects (Table 2.19). Recruitment was positively influenced by percent vine cover ($P < 0.01$, $\beta = 0.017$, SE = 0.005; Figure 2.18a), but even though shrub cover was included in the top model, it did not affect the recruitment rate ($P = 0.13$, $\beta = 0.022$, SE = 0.015). Shrub cover was also included in estimates for apparent survival, but was not significant ($P = 0.71$, $\beta = 0.007$, SE = 0.019). Increasing Julian date decreased detection probability in all models ($P < 0.001$, $\beta = -0.135$, SE = 0.036; Figure 2.18b).

White-eyed Vireo abundance post stand initiation (2011–2021) was greatly influenced by understory vegetation structure, specifically by percentage of vine, shrub, and forbs cover (Table 2.20). Shrub and forbs were in both recruitment and apparent survival formulas, with opposite effects. Increasing percentage of forb cover decreased the recruitment rate ($P < 0.001$, $\beta = -0.066$, SE = 0.019; Figure 2.19a) but increased apparent survival ($P < 0.05$, $\beta = 0.054$, SE = 0.026; Figure 2.19d). Increasing shrub cover positively influenced the recruitment rate ($P < 0.001$, $\beta = 0.031$, SE = 0.006; Figure 2.19b) while decreasing the apparent survival probability ($P < 0.01$, $\beta = -0.075$, SE = 0.023; Figure 2.19e). In addition, increasing vine cover significantly increased the recruitment rate ($P < 0.05$, $\beta = 0.013$, SE = 0.006; Figure 2.19c). Julian date had a positive influence on probability of detection ($P = 0$, $\beta = 0.059$, SE = 0.004; Figure 2.19f); cloud cover, while included in the top model, did not affect the probability of detection ($P = 0.85$, $\beta = 0.000$, SE = 0.001).

Wood Thrush

Wood Thrush abundance ranged from 0 to 344 individuals, with density estimates ranging from 0–7.31 males/hectare. The highest estimations were in stands approaching mid-rotation, prior to thinning, and in late rotation stands 0-6 years pre harvest (Figure 2.12). Mean abundance was 100 individuals post stand initiation and 203 individuals post thinning. Estimations were highest in the Pine Control and Herbicide treated, late rotation stands, but could not be estimated in the post stand initiation Pine Control plots due to the limited number of detections in those plots (Figure 2.13d).

Post thinning, Wood Thrush abundance was significantly influenced by graminoid cover. Increasing graminoid cover decreased the recruitment rate ($P < 0.001$, $\beta = -0.435$, SE = 0.095; Figure 2.20a), while increasing the apparent survival probability ($P < 0.05$, $\beta = 0.679$, SE = 0.300; Table 2.19 and Figure 2.20b). Probability of detection was significantly influenced by Julian date, with detection probabilities increasing in response to increase date ($P < 0.001$, $\beta = 0.191$, SE = 0.055; Figure 2.20c).

Wood Thrush abundance post stand initiation (2011–2021) was significantly influenced by percent vine cover, though shrub ($P = 0.51$, $\beta = -4.140$, SE = 6.24), temperature ($P = 0.10$, $\beta = -0.155$, SE = 0.094), and cloud ($P = 0.36$, $\beta = 0.007$, SE = 0.008) were also included in the top model (Table 2.20). Increasing vine cover caused a decrease in recruitment rate ($P < 0.01$, $\beta = -0.612$, SE = 0.231; Figure 2.21). However, goodness-of-fit testing indicated lack of fit via underdispersion for the top model (c-hat 0.61), indicating uncertainty in the estimations.

Discussion

When considering avian population trends and conservation needs within forest, working or otherwise, there are undeniable benefits to collecting long term data through point count surveys (Collins 2001, Purcell 2011). While issues occur in using count data collected by independent observers of varying abilities, trends estimated at specific time intervals can mitigate issues arising from the use of multiple observers (Link and Sauer 1998). Large, longitudinal data sets can encompass a large amount of variability that might not otherwise be observed in short term studies and can provide a more thorough evaluation of temporal fluctuations in avian populations (Holmes and Sherry 2001, Brooks and Bonter 2010). These records may prove valuable in provoking warnings of undesirable, or beneficial, changes in the environment that significantly impact the bird community. Relatively few studies have the ability to document such changes as few areas exist where long term research is a primary objective. As such, this study had a rare opportunity to examine temporal changes in avian community structure in response to forest management practices within the exact same forest stands.

Species diversity and richness are common measures used to quantify the impacts of habitat manipulations but assessing changes to priority species in response to changing conditions must be a considered by any land manager (Arponen et al. 2005, Veech and Crist

2007, Russell et al. 2009). Richness is an often-used measure of community status, but alternative measures, such as diversity estimates and guild analysis, may prove more useful depending on community composition (Russell et al. 2009). I found changes in species richness dependent on forest stage (forest succession) and treatment, specifically increasing richness in response to the burn plus herbicide treatment for all resident species and early successional/scrub guilds, but a decreasing richness in response to herbicide for neotropical migrants. Hardwood control increased early successional/scrub species richness, but switchgrass intercropping decreased woodland species richness and diversity; both relationships were expected given known habitat associations. However, grouping by stage led to an averaging of yearly impacts, likely decreasing the influence of a singular management action across time. Models using species of conservation concern generally performed poorly when compared to migratory and habitat guild diversity and richness models, as did models investigating the influence of forest stage and treatment on CV. These indices alone also do not consider all species equally and are not able to assess the conservation value of rarities (Arduini and Ercoli 2012). This suggests that the conservation value of each forest stage and treatment cannot be uniformly predicted using traditional methods, and further monitoring and analysis is needed.

These results were unexpected as previous research, while focusing on shorter temporal periods, found that switchgrass intercropping generally did not produce consistent differences in diversity indices with compared to Pine Control plots (Loman 2014, Fuller-Morris 2018). However, earlier research highlighted that this intercropping method produced differences in abundance by species guild, making the inference that abundance changes should lead to diversity changes over longer periods of time (Loman 2014). It is important to recognize that while models incorporate forest stage and treatment across time, previous efforts documenting effects within each year showed variation in response to forest succession (Loman 2014). Dividing species into guilds based on similar life history traits can prove useful in understanding processes occurring within the larger community, particularly in response to management (Verner 1984), but may ignore the value of rare species or species of conservation concern. Species guilds may respond differently to management actions, with large, similarly limited groups of species reflecting larger ecosystem processes (Lambeck 1997). Active management in neighboring stands, while not investigated in this study, likely influenced community composition within the study area. Still, while it can be predicted that switchgrass intercropping will affect migratory and habitat association guilds differently across time, the effects of switchgrass retention in currently unmanaged plots, with closing forest canopies and future mechanical thinning, needs further attention. It is clear that active management can have both positive and negative consequences on avian diversity across a long temporal period. But it should be noted that caution is needed with these metrics between treatments and across years to not infer relative breeding habitat quality from the outcomes (Van Horne 1983).

For many priority species, forest age, as a metric of time since stand initiation, produced a negative correlation within RDA models. But given the lack of fit of both RDA analysis and LMM for species of conservation concern, there is clearly no one management solution for imperiled species. Often abundances and species compositions vary along successional gradients, regularly in response to vegetation structure changes or forestry practices (Wiens and Rotenberry 1981, DeGraaf et al. 1998, Holmes and Sherry 2001). This aligns with the results from this research, emphasizing the importance of considering forest age as a measure of succession in models assessing forest management practices as management may not affect each stage of forest succession in the same way. To further investigate treatment effects, further research is needed comparing thinned versus unthinned stands in response to different practices, as forest age interacting with treatment can become a confounding factor to any analysis.

Perhaps just as intriguing as total community or guild outcomes is the identification of which vegetation groups alter the abundance of specific species across time. Avian habitat specialization in response to specific physical characteristics of the environment (vegetation structure and composition, availability of food resources provided by different plant taxa, etc.) is well documented and discussed (e.g., MacArthur and MacArthur 1961, Weins 1974, Rotenberry 1985, Moorman 2000). Many species of conservation concern are considered habitat specialists, which validates the need to maintain stands of all age classes and vegetation structures within the larger landscape (see Sustainable Forestry Initiative 2022) as the ability to support high population numbers in this system is likely dependent on availability, and quality, of both early successional, open forest vegetation and closed canopy conditions. Impacts of forest succession, regardless of management, are clearly shown in NMDS analysis, but further evaluation into changes in vegetation cover reveals additional effects. For the specific species chosen for analysis within this study, general information regarding documented breeding habitat type, and occasionally conditions, have been described (see Evans et al. 2020, Guzy and Ritchison 2020, McDonald 2020, Hopp 2022), but less is known about specific conditions driving abundance within managed forest systems.

Indeed, for Common Yellowthroat, a common species using many cover types throughout much of North America, Guzy and Ritchison (2020) explicitly note that quantitative studies are lacking, and most aspects of their biology remain unknown. Common Yellowthroat generally responded positively across time to active management techniques, possibly in response to microhabitat selection of within the available cover types and creation of early successional conditions through management across the larger landscape (Mitchell et al. 2001, Campbell et al. 2007), while species such as Wood Thrush are known to be less constrained and selecting habitats and cover types on broader landscape scales (Kilgo et al. 1998, Mitchell et al. 2001). Interestingly, some species classified as early successional/scrub or woodland have an association with similar cover types but respond differently depending on pine growth and time since stand initiation. As an example, both White-eyed Vireo and Kentucky Warbler are known to associate with thick forest understories and forest stands with high levels of structural complexity, but occupancy, and consequently abundance, within stands might be positively or negatively dictated by additional factors such as patchiness and light penetration (Conner et al. 1983, Hodges, Jr. and Krementz 1996, McDonald 2020, Hopp 2022). Canopy cover is known to influence understory vegetation throughout the full stand rotation within managed forests, but the degree to which forest gaps and open canopies influence foraging and nesting site selection for many species is not clearly understood (Kilgo et al. 1996, Reich et al. 2020, Guzy and Ritchinson 2020, McDonald 2020, Price et al. 2020). White-eyed Vireo are frequently classified as mid succession inhabitants, but high abundances have been documented in regenerating pine stand up to ten years post stand initiation (Annand and Thompson 1997, Robinson and Robinson 1999, Twedt et al. 1999, Twedt and Somershoe 2009), a period of time in our study where the canopy is reaching closure and the understory remains dense and complex.

Relatively open conditions were observed at several points in this study, specifically due to mechanically thinning, prescribed fire, and intercropping switchgrass which subsequently delayed shrub and forb growth. Avian diversity typically decreases in response to increasing understory densities, while prescribed fire does promote understory plant diversity and positive effects on bird communities (Greene et al. 2016). Open forest conditions created by prescribed fire that likely promoted understory diversity had a positive impact on diversity of early successional/scrub and grassland species across time in some models. Total avian diversity was higher after thinning but was also high during the mid-rotation years prior to thinning when a closing canopy was observed. One possible explanation for these findings is provided in the treatments used during this time period. Switchgrass intercropping and hardwood control should aid in maintaining open pine conditions due to removal of encroaching hardwoods and inhibiting understory growth (Lane et al. 2011).

Full analysis revealed that active management, with practices used at different times in the stand rotation, had the potential to maintain avian diversity and create conditions for many priority bird species. Structural diversity, created by changing vegetation structure and composition through active management, is recognized as crucial for habitat conservation for countless species (Elliott et al. 1997, Ciancio and Nocentini 2011, Arduini and Ercoli 2012). The addition of multiple management protocols during a full stand rotation may enhance biodiversity, contributing to sustainable forestry practices. Removal of active management techniques (e.g., fire, mechanical thinning) would have allowed a shrubby understory to take hold, thus compromising the many functional and ecological attributes of the managed system (Lindenmayer et al. 2011) by shifting the vegetation structure further from a grassland state after harvest (Hess and Tschinkel 2017). While models documented some negative correlations to certain management techniques, impact estimates were small. These results could be a result of condensing the collected data into smaller sample sizes to create equal comparisons across years or behavioral responses to the changing conditions surrounding each study area. Conservation value was high in early and late stage stands, though given the relatively high yearly species turnover rates in the Young Open Canopy stage as well as the NMDS results, avian community composition and structure was different between these two periods.

With species turnover rates at comparatively higher levels during the first few years following stand initiation, a need existed to document shifts in forest structure during the same time period. In the first few years following stand initiation, when disturbance is high, bird communities exhibit drastically smaller total and individual abundance estimates (Loman 2014). Site preparation intensity has clear impacts on bird communities (Lane et al. 2011, Iglay et al. 2012, Loman 2014, Greene et al. 2016), and further disturbance in subsequent years has the potential to create a lag in time until community convergence. As woody growth increases across time, decreases the ambient temperatures and light level create a general decrease in flora, forb, and grass abundance that in turn alters the forest vegetation structure and food sources for many wildlife species (DeGraaf and Yamasaki 2003, Mathis et al. 2021). While previously explored in Loman (2014), I found that even without consistent collection of vegetation metrics, a change in the percent cover and pine growth dictated the predicted abundance of certain avian species. Fewer vegetation changes occur in late rotation, thinned forests; less forest structure changes lead to decreases in bird community structure differences, moving gradually towards a convergence point between treatments throughout the later years post management (Hobson and Schieck 1999). This phenomenon was clearly observed in the NMDS analysis, showing avian communities in the later years of the pine rotation were similar in structure, with little variation on either axis.

While both NMDS had relatively high R^2 values, the fitted RDA value indicated a lower fit and an increased need for additional forest community metrics. Throughout the 21-year period, metrics such as canopy cover and shrub cover were not consistently collected. Had these metrics been collected across all years, estimations of avian community response to forest properties could be assessed in more detail. Conclusions about avian habitat selection in response to management decisions can be closely related to selected attributes of habitats, highlighting the importance of carefully choosing survey components for measurement (Rice et al. 1984). Many long-term studies, including this research, do not follow the exact same protocols each year, potentially complicating comparative efforts (Hall 1984, Leck et al. 1988, Brooks and Bonter 2010). Focusing on standardizing protocols when creating and continuing long term projects, specifically when estimating demographic parameters and community response to succession, should be a priority for any future research (Robinson et al. 2009).

Conclusion and Management Implication

This research provides support for monitoring avian community changes years after active management has taken place. This work is a continuation of previous studies on the everchanging bird community within a managed pine forest in Mississippi in response to forest management but was the first to use the collected 21-years' worth of data. It shows clear impacts of specific silvicultural techniques that have implications on the presence or absence, and consequently abundance, of particular bird species, possibly dependent on each species sensitivity to anthropogenic actions altering vegetation structure and composition. But care needs to be taken as the documented measurements and metrics may not produce direct cause and effect relationships, nor be indicative of habitat quality and subsequently annual productivity. Both positive and negative impacts of specific management actions may not have been able to have been fully quantified in this study, as a longer period post treatment and additional forest measurements are likely required for certain species (Wiens and Rotenberry 1985, Russell et al. 2009).

Future research should focus on understanding how the avian community within this system is impacted by spatial arrangement of treatments, impacts of adjacent harvests, and the application of management treatments at different time periods during the stand rotation. This research should consider the possible effects of prior management, post-harvest and stand initiation, to investigate if use of prescribed fire in the previous rotation influences vegetation growth post initiation. In addition, adding studies focused on evaluating reproductive success and survivorship in response to forest succession and management across time should be considered. When working towards meeting conservation goals, a range of practices can be used to maintain avian biodiversity. Landowners should use all tools available to them that would aid in meeting management and conservation goals, including the use of prescribed fire, with or without herbicide, at regular intervals. Full understanding of potential impacts of implemented forest management practices within large forested systems requires knowledge of prevalence and distribution of similar treatments within the landscape, and the potential interactions with additional forest conditions contained within the larger forested system. To recognize the full potential of pine forests to meet conservation needs for many bird species, further coordinated efforts are needed at multiple scales.

Tables and Figures

Table 2.1 Timetable of avian and vegetation surveys and active management actions, from 1999–2021 in 5 managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi. Activity is marked with an “X”.

Year	Forest Stage	Avian Surveys	Vegetation Surveys	Clearcut Harvest	Stand Initiation	Herbicide	Burn	Hardwood Control	Switchgrass Intercrop
1999	Late Stage	X	X			X			
2000	Late Stage	X	X				X		
2001	Late Stage	X	X						
2002	Late Stage	X	X						
2003	Late Stage	X	X				X		
2004	Late Stage	X	X						
2005	Late Stage	X	X						
2006	Late Stage	X	X				X		
2007	Late Stage	X	X						
2008	Late Stage	X	X						
2009				X					
2010					X				
2011	Young Open Canopy	X	X						X
2012	Young Open Canopy	X	X						X
2013	Young Open Canopy	X	X						X
2014	Young Open Canopy	X	X						X
2015	Young Open Canopy	X	X					X	X
2016	Mid Stage	X	X						
2017	Mid Stage	X	X						
2018	Mid Stage	X	X						
2019	Mid Stage	X	X						
2020	Mid Stage	X	X						
2021	Mid Stage	X	X						

Table 2.2 List of breeding avian species, including flyovers, detected during point count surveys May-June 1999–2008 and 2011–2021 within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi, with the associated American Ornithological Society/Institute for Bird Populations (AOU) alpha code (Pyle and DeSante 2003), modified Partners in Flight (PIF) ranks, migratory status, and associated habitat.

Species		AOU Code	PIF Score[a]	Migratory Status[b]	Habitat[c]	Detected 1999–2008	Detected 2011–2021
Acadian Flycatcher	*Empidonax virescens*	ACFL	2	N	W	X	X
American Crow	*Corvus brachyrhynchos*	AMCR	1	R	W	X	X
American Goldfinch	*Spinus tristis*	AMGO	1	S	ES/S		X
American Kestrel*	*Falco sparverius*	AMKE	3	R	G	X	X
Bachman's Sparrow*	*Peucaea aestivalis*	BACS	4	R	W	X	
Barn Swallow	*Hirundo rustica*	BARS	1	N	U		X
Barred Owl	*Strix varia*	BARO	1	R	W		X
Black Vulture	*Coragyps atratus*	BLVU	3	R	W	X	X
Black-and-white Warbler	*Mniotilta varia*	BAWW	1	N	W	X	X
Blue Grosbeak	*Passerina caerulea*	BLGR	1	N	ES/S	X	X
Blue Jay	*Cyanocitta cristata*	BLJA	1	S	ES/S	X	X
Blue-gray Gnatcatcher	*Polioptila caerulea*	BGGN	1	N	W	X	X
Broad-winged Hawk	*Buteo platypterus*	BWHA	1	S	W	X	X
Brown-headed Cowbird	*Molothrus afer*	BHCO	1	S	G	X	X
Brown-headed Nuthatch	*Sitta pusilla*	BHNU	3	R	W	X	X
Brown Thrasher	*Toxostoma rufum*	BRTH	2	N	ES/S	X	X
Carolina Chickadee	*Poecile carolinensis*	CACH	1	R	W	X	X
Carolina Wren	*Thryothorus ludovicianus*	CARW	1	R	ES/S	X	X
Chimney Swift	*Chaetura pelagica*	CHSW	4	N	U		X
Chipping Sparrow	*Spizella passerina*	CHSP	1	R	U		X
Chuck-will's-widow*	*Antrostomus carolinensis*	CWWI	3	N	W		X
Common Grackle	*Quiscalus quiscula*	COGR	1	S	U		X
Common Ground-Dove	*Columbina passerina*	CGDO	2	R	ES/S		X
Common Nighthawk	*Chordeiles minor*	CONI	3	N	G		X
Common Yellowthroat	*Geothlypis trichas*	COYE	1	N	ES/S		X
Cooper's Hawk	*Accipiter cooperii*	COHA	1	N	W	X	X
Downy Woodpecker	*Dryobates pubescens*	DOWO	1	R	W	X	X
Eastern Bluebird	*Sialia sialis*	EABL	1	R	G	X	X
Eastern Kingbird*	*Tyrannus tyrannsu*	EAKI	3	N	G	X	X
Eastern Meadowlark*	*Sturnella magna*	EAME	3	S	G	X	X
Eastern Phoebe	*Sayornis phoebe*	EAPH	1	R	W		X
Eastern Towhee*	*Pipilo erythrophthalmus*	EATO	3	S	ES/S	X	X
Eastern Wood-Pewee*	*Contopus virens*	EAWP	3	N	W	X	X
Field Sparrow*	*Spizella pusilla*	FISP	3	S	ES/S		X
Fish Crow	*Corvus ossifragus*	FICR	1	R	WE	X	X
Gray Catbird	*Dumetella carolinensis*	GRCA	1	N	ES/S	X	X
Great Crested Flycatcher	*Myiarchus crinitus*	GCFL	1	N	W	X	X
Great Horned Owl	*Bubo virginianus*	GHOW	1	R	W		X

Table 2.2 (continued)

Species		AOU Code	PIF Score[a]	Migratory Status[b]	Habitat[c]	Detected 1999–2008	Detected 2011–2021
Hairy Woodpecker	*Dryobates villosus*	HAWO	1	R	W	X	X
Hooded Warbler	*Setophaga citrina*	HOWA	1	N	W	X	X
Indigo Bunting*	*Passerina cyanea*	INBU	4	N	ES/S	X	X
Kentucky Warbler*	*Geothlypis formosa*	KEWA	4	N	W	X	X
Killdeer	*Charadrius vociferus*	KILL	1	S	G		X
Mourning Dove	*Zenaida macroura*	MODO	1	S	U	X	X
Northern Bobwhite*	*Colinus virginianus*	NOBO	4	R	ES/S	X	X
Northern Cardinal	*Cardinalis cardinalis*	NOCA	1	R	ES/S	X	X
Northern Flicker	*Colaptes auratus*	NOFL	2	S	W	X	X
Northern Mockingbird	*Mimus polyglots*	NOMO	1	R	U		X
Northern Parula	*Setophaga americana*	NOPA	1	N	W	X	X
Orchard Oriole	*Icterus spurius*	OROR	3	N	W		X
Ovenbird	*Seiurus aurocapilla*	OVEN	2	N	W		X
Painted Bunting*	*Passerina ciris*	PABU	2	N	ES/S	X	
Pileated Woodpecker	*Dryocopus pileatus*	PIWO	1	R	W	X	X
Pine Warbler	*Setophaga pinus*	PIWA	1	R	W	X	X
Prairie Warbler*	*Setophaga discolor*	PRAW	3	N	ES/S	X	X
Purple Martin	*Progne subis*	PUMA	3	N	U		X
Red-bellied Woodpecker	*Melanerpes carolinus*	RBWO	1	R	W	X	X
Red-eyed Vireo	*Vireo olivaceus*	REVI	1	N	W	X	X
Red-headed Woodpecker*	*Melanerpes erythrocephalus*	RHWO	3	R	W	X	X
Red-shouldered Hawk	*Buteo lineatus*	RSHA	1	S	W	X	X
Red-tailed Hawk	*Buteo jamaicensis*	RTHA	1	S	W	X	X
Ruby-throated Hummingbird	*Archilochus colubris*	RTHU	1	N	W	X	X
Scarlet Tanager	*Piranga olivacea*	SCTA	2	N	W	X	X
Summer Tanager	*Piranga rubra*	SUTA	1	N	W	X	X
Swainson’s Warbler*	*Limnothlypis swainsonii*	SWWA	1	N	W		X
Tufted Titmouse	*Baeolophus bicolor*	TUTI	1	R	W	X	X
Turkey Vulture	*Cathartes aura*	TUVU	1	R	W	X	X
White-eyed Vireo	*Vireo griseus*	WEVI	1	N	ES/S	X	X
Wild Turkey	*Meleagris gallopavo*	WITU	1	R	W	X	X
Wood Thrush*	*Hylocichla mustelina*	WOTH	4	N	W	X	X
Worm-eating Warbler*	*Helmitheros vermivorum*	WEWA	2	N	W	X	X
Yellow-billed Cuckoo*	*Coccyzus americanus*	YBCU	3	N	W	X	X
Yellow-breasted Chat	*Icteria virens*	YBCH	1	N	ES/S	X	X
Yellow-throated Vireo	*Vireo flavifrons*	YTVI	1	N	W	X	X
Yellow-throated Warbler	*Setophaga dominica*	YTWA	1	N	W	X	X

*Denotes priority bird classification by the Eastern Gulf Coastal Plain Joint Venture Landbird Conservation Plan (Greene et al. 2021)

[a] PIF Rank calculated from the Partners in Flight Avian Conservation Assessment Database using the methodology of Nuttle et al. (2003)

[b] Migratory status abbreviations: N- neotropical migrant, S- short distance migrant, R- resident; retrieved from the Breeding Bird Survey database (Sauer et al. 2017)

[c] Associated habitat type abbreviations: ES/S - early-successional/scrub, G - grassland, U - urban, WE - wetland, W - woodland; retrieved from the Breeding Bird Survey database (Sauer et al. 2017)

Table 2.3 The set of dynamic (recruitment-survival) abundance models fit to avian point count data within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi, in May and June 1999–2008. Additive covariate relationships are indicated by + (i.e., shrub + forbs). Model selection was based on Akaike's Information Criterion (AIC). Models were ranked by AICc, and the number of parameters (K), ΔAICc, model weight (W_i), cumulative model weight (Cum. W_i), and model log likelihood (LogLik) were reported.

Species		**Model**	**K**	**AICc**	**ΔAICc**	**W_i**	**Cum.W_i**	**LogLik**
Common Yellowthroat	1	~λ(.) ~γ(canopy + shrub + vine) ~ω(forbs + shrub) ~p(jdate)	10	1944.89	0.00	0.60	0.6	-962.4463
	2	~λ(.) ~γ(canopy + shrub + vine) ~ω(forbs) ~p(jdate)	9	1945.72	0.82	0.40	1.00	-963.8579
	3	~λ(.) ~γ(.) ~ω(.) ~p(jdate)	5	2058.79	113.90	0.00	1.00	-1024.396
	4	~λ(.) ~γ(.) ~ω(.) ~p(.)	4	2282.87	337.98	0.00	1.00	-1137.435
Kentucky Warbler	1	~λ(.) ~γ(graminoid + shrub) ~ω(.) ~p(jdate)	7	2574.69	0.00	0.22	0.22	-1280.35
	2	~λ(.) ~γ(graminoid + forbs + shrub) ~ω(.) ~p(jdate)	8	2574.85	0.16	0.20	0.42	-1279.43
	3	~λ(.) ~γ(graminoid + forbs) ~ω(shrub) ~p(jdate)	8	2574.91	0.22	0.2	0.61	-1279.46
	4	~λ(.) ~γ(graminoid) ~ω(shrub) ~p(jdate)	7	2575.27	0.58	0.16	0.78	-1280.64
	5	~λ(.) ~γ(graminoid + shrub) ~ω(shrub) ~p(jdate)	8	2575.94	1.25	0.12	0.89	-1279.97
	6	~λ(.) ~γ(graminoid + forbs + shrub) ~ω(shrub) ~p(jdate)	9	2576.12	1.43	0.11	1.00	-1279.06
	7	~λ(.) ~γ(.) ~ω(.) ~p(jdate)	5	2620.89	46.2	0.00	1.00	-1205.44
	8	~λ(.) ~γ(.) ~ω(.) ~p(.)	4	2921.65	346.96	0.00	1.00	-1456.83
White-eyed Vireo	1	~λ(.) ~γ(vine + shrub) ~ω(shrub) ~p(jdate)	8	3906.04	0.00	0.43	0.43	-1945.02
	2	~λ(.) ~γ(vine) ~ω(shrub) ~p(jdate)	7	3906.18	0.13	0.41	0.84	-1946.09
	3	~λ(.) ~γ(vine + shrub) ~ω(shrub + forbs) ~p(jdate)	9	3908.04	2.00	0.16	1.00	-1945.02
	4	~λ(.) ~γ(.) ~ω(.) ~p(jdate)	5	3929.23	23.19	0.00	1.00	-1954.9
	5	~λ(.) ~γ(.) ~ω(.) ~p(.)	4	4325.5	419.46	0.00	1.00	-2158.75

Table 2.3 (continued)

Species		Model	K	AICc	ΔAICc	W_i	Cum.W_i	LogLik
Wood Thrush	1	~λ(.) ~γ(graminoid) ~ω(graminoid) ~p(jdate)	7	1742.27	0.00	0.36	0.36	-864.13
	2	λ(.) ~γ(graminoid + forbs + vine) ~ω(graminoid) ~p(jdate)	9	1742.95	0.68	0.26	0.61	-862.47
	3	~λ(.) ~γ(graminoid + forbs) ~ω(graminoid) ~p(jdate)	8	1743.47	1.20	0.20	0.81	-863.73
	4	~λ(.) ~γ(graminoid + vine) ~ω(graminoid) ~p(jdate)	8	1743.55	1.28	0.19	1.00	-863.77
	5	~λ(.) ~γ(.) ~ω(.) ~p(jdate)	6	1789.41	47.14	0.00	1.00	-888.04
	6	~λ(.) ~γ(.) ~ω(.) ~p(.)	5	1898.35	156.08	0.00	1.00	-945.17

λ – lambda; formula for initial abundance
γ – gamma; formula for recruitment rate
ω – omega; formula for apparent survival probability
p – formula for detection probability

Table 2.4 The set of dynamic (recruitment-survival) abundance models fit to avian point count data within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi, in May and June 2011–2021. Additive covariate relationships are indicated by + (i.e., shrub + forbs). Model selection was based on Akaike's Information Criterion (AIC). Models were ranked by AICc, and the number of parameters (K), ΔAICc, model weight (W_i), cumulative model weight (Cum. W_i), and model log likelihood (LogLik) were reported.

Species		Model	K	AICc	ΔAICc	W_i	Cum.W_i	LogLik
Common Yellowthroat	1	~λ(.) ~γ(allgrass + vine) ~ω(vine) ~p(cloud + temp + jdate)	10	2825.31	0.00	0.31	0.31	-1402.54
	2	~λ(.) ~γ(allgrass + vine) ~ω(vine) ~p(temp + jdate)	9	2823.32	0.01	0.31	0.61	-1403.66
	3	~λ(.) ~γ(allgrass) ~ω(vine) ~p(temp + jdate)	8	2826.12	0.81	0.20	0.82	-1405.06
	4	~λ(.) ~γ(allgrass) ~ω(vine) ~p(cloud + temp + jdate)	9	2826.36	1.05	0.18	1.00	-1404.18
	5	~λ(.) ~γ(.) ~ω(.) ~p(cloud + temp + jdate)	7	3169.88	344.57	0.00	1.00	-1577.94
	6	~λ(.) ~γ(.) ~ω(.) ~p(.)	4	3461.85	636.54	0.00	1.00	1726.93
Kentucky Warbler	1	~λ(.) ~γ(allgrass + shrub) ~ω(.) ~p(cloud + jdate)	8	682.10	0.00	0.87	0.87	-331.03
	2	~λ(.) ~γ(allgrass + shrub) ~ω(.) ~p(jdate)	7	685.80	3.70	0.13	1.00	-334.38
	3	~λ(.) ~γ(.) ~ω(.) ~p(cloud + jdate)	6	708.85	26.75	0.00	1.00	-348.43
	4	~λ(.) ~γ(.) ~ω(.) ~p(.)	4	716.59	34,49	0.00	1.00	-354.29
White-eyed Vireo	1	~λ(.) ~γ(vine + shrub+forbs) ~ω(shrub + forbs) ~p(cloud + jdate)	11	2850.57	0.00	0.66	0.66	-1414.28
	2	~λ(.) ~γ(vine + shrub + forbs) ~ω(vine + shrub + forbs) ~p(cloud + jdate)	12	2851.90	1.34	0.34	1.00	-1413.95
	3	~λ(.) ~γ(.) ~ω(.) ~p(cloud + jdate)	6	3067.02	216.45	0.00	1.00	-1527.51
	4	~λ(.) ~γ(.) ~ω(.) ~p(.)	4	3385.97	535.40	0.00	1.00	-1688.99

Table 2.3 (continued)

Species		Model	K	AICc	ΔAICc	W_i	Cum.W_i	LogLik
Wood Thrush	1	~λ(.) ~γ(vine) ~ω(shrub) ~p(cloud + temp)	8	134.61	0.00	0.52	0.52	-59.31
	2	~λ(.) ~γ(vine) ~ω(.) ~p(cloud + temp)	7	136.39	1.78	0.22	0.74	-61.20
	3	~λ(.) ~γ(shrub + vine) ~ω(shrub + vine) ~p(cloud + temp)	10	136.79	2.18	0.18	0.92	-58.39
	4	~λ(.) ~γ(.) ~ω(.) ~p(cloud + temp)	6	138.14	3.53	0.04	0.96	-63.07
	5	~λ(.) ~γ(shrub + vine) ~ω() ~p(cloud + temp)	8	138.29	3.68	0.04	1.00	-61.15
	6	~λ(.) ~γ(.) ~ω(.) ~p(.)	4	151.35	16.74	0.00	1.00	-71.68

λ – lambda; formula for initial abundance
γ – gamma; formula for recruitment rate
ω – omega; formula for apparent survival probability
p – formula for detection probability

Table 2.5 Species richness by forest age, stage, and treatment during point count surveys May-June 1999–2008 and 2011–2021 within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. Data were not collected in stand ages 12-17 years since stand initiation.

Forest Age (years)	Forest Stage	Treatment						Total
		Burn	Burn + Herbicide	Pine Control	Hardwood Control	Herbicide	Switchgrass Intercropped	
18	Late Stage	28	33	31		33		39
19	Late Stage	33	38	35		36		44
20	Late Stage	34	33	29		34		43
21	Late Stage	31	30	31		30		34
22	Late Stage	30	33	29		32		40
23	Late Stage	44	39	36		31		49
24	Late Stage	35	36	34		32		41
25	Late Stage	32	31	38		30		45
26	Late Stage	31	33	29		28		40
27	Late Stage	31	33	34		30		41
1	Young Open Canopy			27			15	30
2	Young Open Canopy			33			22	36
3	Young Open Canopy			29			23	29
4	Young Open Canopy			30			22	33
5	Young Open Canopy			26			23	28
6	Mid Stage			35	31		33	39
7	Mid Stage			33	33		30	37
8	Mid Stage			31	33		30	39
9	Mid Stage			36	36		38	45
10	Mid Stage			35	40		41	50
11	Mid Stage			35	36		31	41

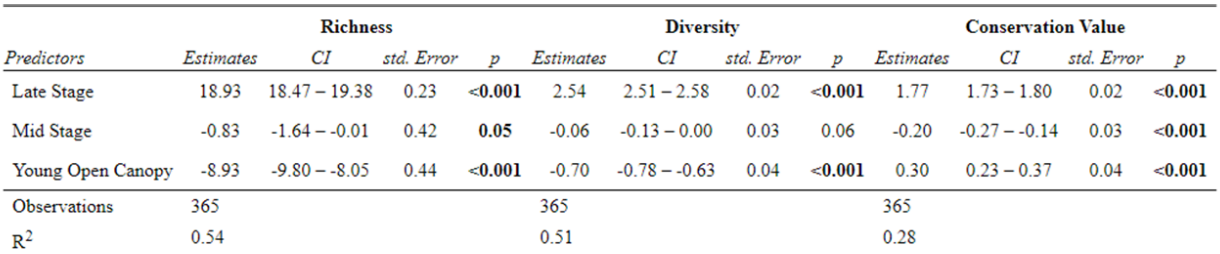
Table 2.6 Diversity indices by forest age, stage, and treatment, from point count surveys May-June 1999–2008 and 2011–2021 within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. Data were not collected in stand ages 12-17 years since stand initiation.

Forest Age (years)	Forest Stage	Treatment						Average
		Burn	Burn + Herbicide	Pine Control	Hardwood Control	Herbicide	Switchgrass Intercropped	
18	Late Stage	2.73	2.78	2.80		2.84		2.79
19	Late Stage	2.80	2.82	2.92		2.84		2.84
20	Late Stage	2.74	2.60	2.78		2.77		2.73
21	Late Stage	2.73	2.67	2.79		2.66		2.71
22	Late Stage	2.73	2.58	2.71		2.70		2.68
23	Late Stage	2.92	2.85	2.86		2.75		2.84
24	Late Stage	2.92	2.83	2.79		2.81		2.84
25	Late Stage	2.76	2.56	2.80		2.65		2.70
26	Late Stage	2.87	2.63	2.80		2.66		2.74
27	Late Stage	2.95	2.81	2.86		2.83		2.86
1	Young Open Canopy			2.63			2.49	2.56
2	Young Open Canopy			2.55			2.58	2.57
3	Young Open Canopy			2.34			2.27	2.30
4	Young Open Canopy			2.35			2.21	2.28
5	Young Open Canopy			2.25			2.22	2.23
6	Mid Stage			2.75	2.75		2.74	2.75
7	Mid Stage			2.76	2.75		2.59	2.70
8	Mid Stage			2.59	2.58		2.41	2.52
9	Mid Stage			2.67	2.60		2.65	2.64
10	Mid Stage			2.89	2.89		2.95	2.91
11	Mid Stage			3.02	2.96		2.84	2.94

Table 2.7 Linear mixed effects model results using treatment and forest stage as fixed effects and survey block as a random effect on species richness, diversity, and Conservation Value (CV) within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. Bolded P-values indicate statistical significance (α = 0.05).

	Richness				**Diversity**				**Conservation Value**			
Predictors	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*
Pine Control	15.57	14.72 – 16.41	0.43	**<0.001**	2.29	2.23 – 2.36	0.03	**<0.001**	1.77	1.69 – 1.85	0.04	**<0.001**
Burn	3.67	2.21 – 5.13	0.74	**<0.001**	0.30	0.18 – 0.42	0.06	**<0.001**	0.05	-0.05 – 0.15	0.05	0.34
Burn+Herbicide	4.01	2.55 – 5.47	0.74	**<0.001**	0.21	0.09 – 0.33	0.06	**<0.001**	0.16	0.06 – 0.26	0.05	**<0.001**
Hardwood Control	1.40	-0.26 – 3.07	0.85	0.10	0.13	-0.01 – 0.26	0.07	0.07	-0.18	-0.29 – -0.06	0.06	**<0.001**
Herbicide	2.47	1.01 – 3.93	0.74	**<0.001**	0.22	0.11 – 0.34	0.06	**<0.001**	-0.05	-0.15 – 0.05	0.05	0.31
Switchgrass Intercropped	-1.37	-2.78 – 0.04	0.72	0.06	-0.15	-0.26 – -0.03	0.06	**0.01**	0.03	-0.07 – 0.12	0.05	0.57
Random Effects												
σ^2	19.68				0.13				0.09			
τ_{00}	0.14_{Stand}				0.00_{Stand}				0.00_{Stand}			
ICC	0.01				0.00				0.05			
N	5_{Stand}				5_{Stand}				5_{Stand}			
Observations	365				365				365			
Marginal R^2 / Conditional R^2	0.15 / 0.16				0.14 / 0.14				0.07 / 0.11			

Table 2.8 Generalized linear model (GLM) results comparing species richness, Shannon-Weaver's diversity index and Conservation Value (CV) by forest age from within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. Bolded P-values indicate statistical significance (α = 0.05).

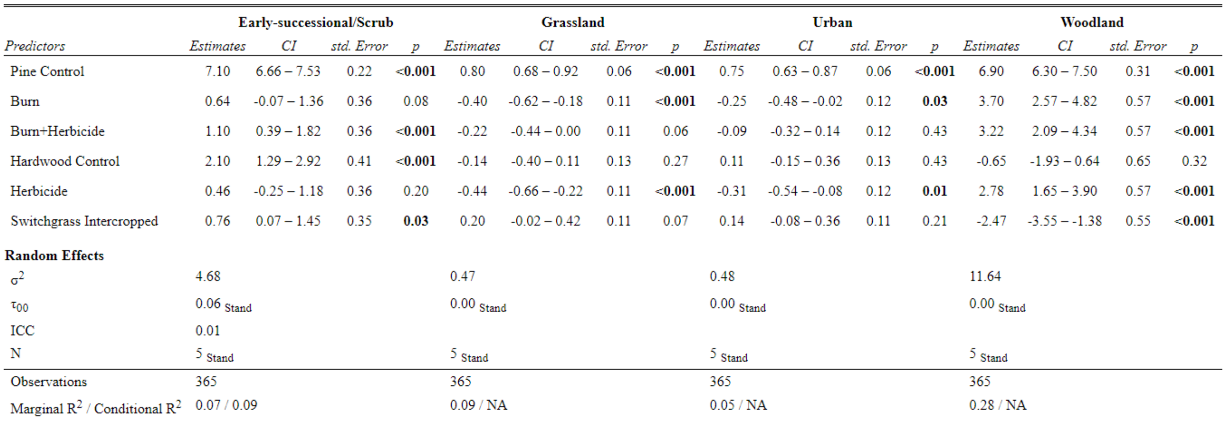


	Richness				**Diversity**				**Conservation Value**			
Predictors	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*
Late Stage	18.93	18.47 – 19.38	0.23	**<0.001**	2.54	2.51 – 2.58	0.02	**<0.001**	1.77	1.73 – 1.80	0.02	**<0.001**
Mid Stage	-0.83	-1.64 – -0.01	0.42	**0.05**	-0.06	-0.13 – 0.00	0.03	0.06	-0.20	-0.27 – -0.14	0.03	**<0.001**
Young Open Canopy	-8.93	-9.80 – -8.05	0.44	**<0.001**	-0.70	-0.78 – -0.63	0.04	**<0.001**	0.30	0.23 – 0.37	0.04	**<0.001**
Observations	365				365				365			
R^2	0.54				0.51				0.28			

Table 2.9 Turnover rates estimating proportion of species turnover between each year and the prior year from within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. Turnover rate was not estimate for 2011 as each stand was recently initiated, and no younger age was available for surveying. Estimate for age 18 used age 11 as the prior age.

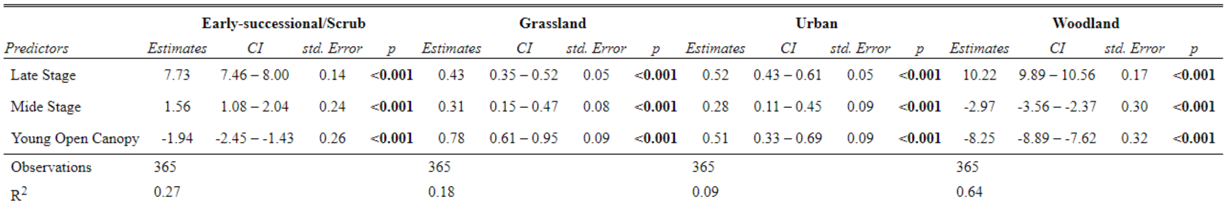
Forest Age	Forest Stage	Year	Turnover Rate
18	Late Stage	2000	0.16
19	Late Stage	2001	0.09
20	Late Stage	2002	0.23
21	Late Stage	2003	0.20
22	Late Stage	2004	0.22
23	Late Stage	2005	0.27
24	Late Stage	2006	0.24
25	Late Stage	2007	0.27
26	Late Stage	2008	0.24
2	Young Open Canopy	2012	0.30
3	Young Open Canopy	2013	0.31
4	Young Open Canopy	2014	0.32
5	Young Open Canopy	2015	0.35
6	Mid Stage	2016	0.37
7	Mid Stage	2017	0.19
8	Mid Stage	2018	0.36
9	Mid Stage	2019	0.38
10	Mid Stage	2020	0.26
11	Mid Stage	2021	0.26

Table 2.10 Linear mixed effects model results using treatment as a fixed effect and survey block as a random effect on species richness for habitat guilds within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. Bolded P-values indicate statistical significance (α = 0.05).



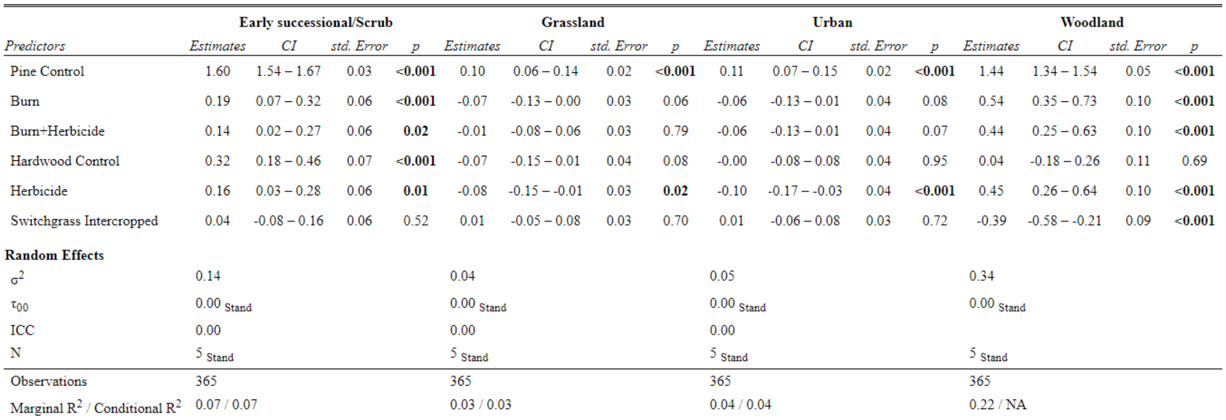
	Early-successional/Scrub				**Grassland**				**Urban**				**Woodland**			
Predictors	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*
Pine Control	7.10	6.66 – 7.53	0.22	<**0.001**	0.80	0.68 – 0.92	0.06	<**0.001**	0.75	0.63 – 0.87	0.06	<**0.001**	6.90	6.30 – 7.50	0.31	<**0.001**
Burn	0.64	-0.07 – 1.36	0.36	0.08	-0.40	-0.62 – -0.18	0.11	<**0.001**	-0.25	-0.48 – -0.02	0.12	**0.03**	3.70	2.57 – 4.82	0.57	<**0.001**
Burn+Herbicide	1.10	0.39 – 1.82	0.36	<**0.001**	-0.22	-0.44 – 0.00	0.11	0.06	-0.09	-0.32 – 0.14	0.12	0.43	3.22	2.09 – 4.34	0.57	<**0.001**
Hardwood Control	2.10	1.29 – 2.92	0.41	<**0.001**	-0.14	-0.40 – 0.11	0.13	0.27	0.11	-0.15 – 0.36	0.13	0.43	-0.65	-1.93 – 0.64	0.65	0.32
Herbicide	0.46	-0.25 – 1.18	0.36	0.20	-0.44	-0.66 – -0.22	0.11	<**0.001**	-0.31	-0.54 – -0.08	0.12	**0.01**	2.78	1.65 – 3.90	0.57	<**0.001**
Switchgrass Intercropped	0.76	0.07 – 1.45	0.35	**0.03**	0.20	-0.02 – 0.42	0.11	0.07	0.14	-0.08 – 0.36	0.11	0.21	-2.47	-3.55 – -1.38	0.55	<**0.001**
Random Effects																
σ^2	4.68				0.47				0.48				11.64			
τ_{00}	0.06_{Stand}				0.00_{Stand}				0.00_{Stand}				0.00_{Stand}			
ICC	0.01															
N	5_{Stand}				5_{Stand}				5_{Stand}				5_{Stand}			
Observations	365				365				365				365			
Marginal R^2 / Conditional R^2	0.07 / 0.09				0.09 / NA				0.05 / NA				0.28 / NA			

Table 2.11 Linear model results comparing species richness for habitat guilds between forest stages within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. Bolded P-values indicate statistical significance (α = 0.05).



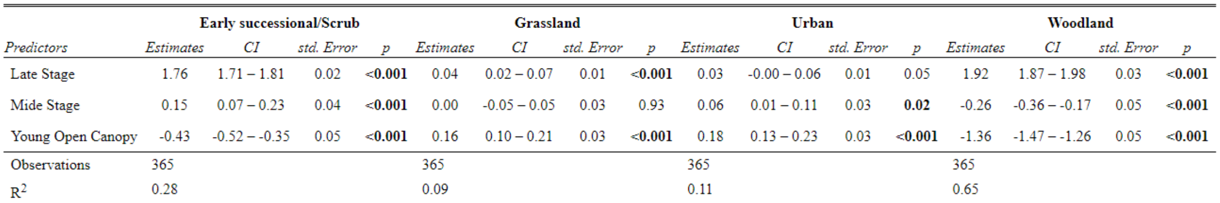
	Early-successional/Scrub				**Grassland**				**Urban**				**Woodland**			
Predictors	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*
Late Stage	7.73	7.46 – 8.00	0.14	**<0.001**	0.43	0.35 – 0.52	0.05	**<0.001**	0.52	0.43 – 0.61	0.05	**<0.001**	10.22	9.89 – 10.56	0.17	**<0.001**
Mide Stage	1.56	1.08 – 2.04	0.24	**<0.001**	0.31	0.15 – 0.47	0.08	**<0.001**	0.28	0.11 – 0.45	0.09	**<0.001**	-2.97	-3.56 – -2.37	0.30	**<0.001**
Young Open Canopy	-1.94	-2.45 – -1.43	0.26	**<0.001**	0.78	0.61 – 0.95	0.09	**<0.001**	0.51	0.33 – 0.69	0.09	**<0.001**	-8.25	-8.89 – -7.62	0.32	**<0.001**
Observations	365				365				365				365			
R^2	0.27				0.18				0.09				0.64			

Table 2.12 Linear mixed effects model results using treatment as a fixed effect and survey block as a random effect on diversity indices for habitat guilds within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. Bolded P-values indicate statistical significance (α = 0.05).



	Early successional/Scrub				**Grassland**				**Urban**				**Woodland**			
Predictors	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*
Pine Control	1.60	1.54 – 1.67	0.03	**<0.001**	0.10	0.06 – 0.14	0.02	**<0.001**	0.11	0.07 – 0.15	0.02	**<0.001**	1.44	1.34 – 1.54	0.05	**<0.001**
Burn	0.19	0.07 – 0.32	0.06	**<0.001**	-0.07	-0.13 – 0.00	0.03	0.06	-0.06	-0.13 – 0.01	0.04	0.08	0.54	0.35 – 0.73	0.10	**<0.001**
Burn+Herbicide	0.14	0.02 – 0.27	0.06	**0.02**	-0.01	-0.08 – 0.06	0.03	0.79	-0.06	-0.13 – 0.01	0.04	0.07	0.44	0.25 – 0.63	0.10	**<0.001**
Hardwood Control	0.32	0.18 – 0.46	0.07	**<0.001**	-0.07	-0.15 – 0.01	0.04	0.08	-0.00	-0.08 – 0.08	0.04	0.95	0.04	-0.18 – 0.26	0.11	0.69
Herbicide	0.16	0.03 – 0.28	0.06	**0.01**	-0.08	-0.15 – -0.01	0.03	**0.02**	-0.10	-0.17 – -0.03	0.04	**<0.001**	0.45	0.26 – 0.64	0.10	**<0.001**
Switchgrass Intercropped	0.04	-0.08 – 0.16	0.06	0.52	0.01	-0.05 – 0.08	0.03	0.70	0.01	-0.06 – 0.08	0.03	0.72	-0.39	-0.58 – -0.21	0.09	**<0.001**
Random Effects																
σ^2	0.14				0.04				0.05				0.34			
τ_{00}	0.00 $_{\text{Stand}}$				0.00 $_{\text{Stand}}$				0.00 $_{\text{Stand}}$				0.00 $_{\text{Stand}}$			
ICC	0.00				0.00				0.00							
N	5 $_{\text{Stand}}$				5 $_{\text{Stand}}$				5 $_{\text{Stand}}$				5 $_{\text{Stand}}$			
Observations	365				365				365				365			
Marginal R^2 / Conditional R^2	0.07 / 0.07				0.03 / 0.03				0.04 / 0.04				0.22 / NA			

Table 2.13 Linear model results comparing diversity indices for habitat guilds between forest stages within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. Bolded P-values indicate statistical significance (α = 0.05).

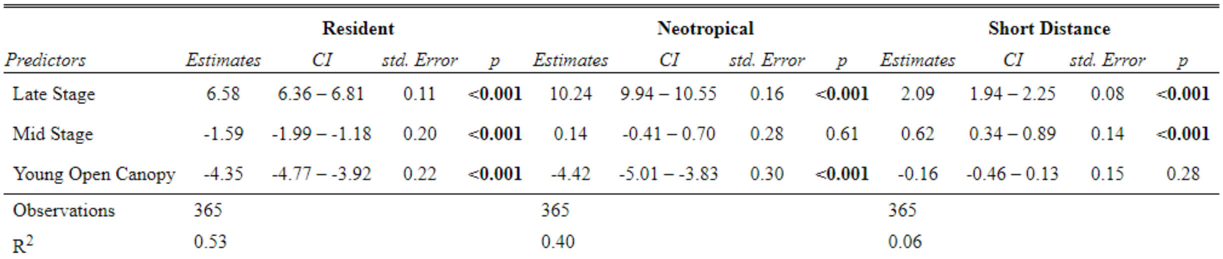


	Early successional/Scrub				**Grassland**				**Urban**				**Woodland**			
Predictors	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*
Late Stage	1.76	1.71 – 1.81	0.02	<**0.001**	0.04	0.02 – 0.07	0.01	<**0.001**	0.03	-0.00 – 0.06	0.01	0.05	1.92	1.87 – 1.98	0.03	<**0.001**
Mide Stage	0.15	0.07 – 0.23	0.04	<**0.001**	0.00	-0.05 – 0.05	0.03	0.93	0.06	0.01 – 0.11	0.03	**0.02**	-0.26	-0.36 – -0.17	0.05	<**0.001**
Young Open Canopy	-0.43	-0.52 – -0.35	0.05	<**0.001**	0.16	0.10 – 0.21	0.03	<**0.001**	0.18	0.13 – 0.23	0.03	<**0.001**	-1.36	-1.47 – -1.26	0.05	<**0.001**
Observations	365				365				365				365			
R^2	0.28				0.09				0.11				0.65			

Table 2.14 Linear mixed effects model results using treatment as a fixed effect and survey block as a random effect on species richness for migration guilds within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. Bolded P-values indicate statistical significance (α = 0.05).

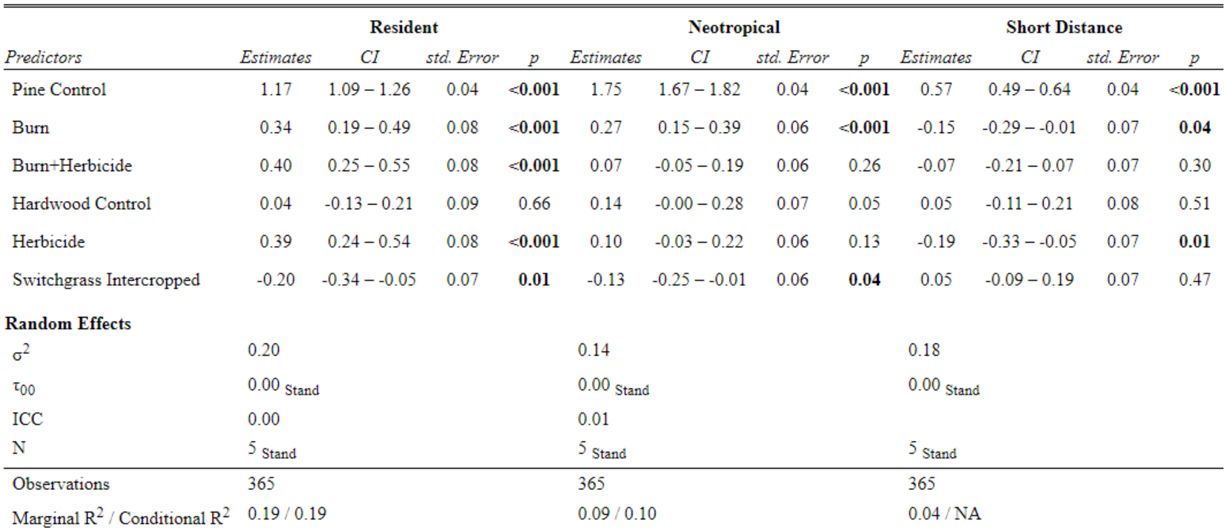
	Resident				**Neotropical**				**Short Distance**			
Predictors	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*
Pine Control	4.56	4.17 – 4.95	0.20	**<0.001**	8.82	8.26 – 9.37	0.28	**<0.001**	2.19	1.99 – 2.39	0.10	**<0.001**
Burn	1.80	1.14 – 2.46	0.33	**<0.001**	2.06	1.17 – 2.96	0.46	**<0.001**	-0.19	-0.57 – 0.18	0.19	0.32
Burn+Herbicide	2.60	1.94 – 3.26	0.33	**<0.001**	1.24	0.35 – 2.14	0.46	**0.01**	0.17	-0.21 – 0.54	0.19	0.38
Hardwood Control	-0.10	-0.86 – 0.65	0.38	0.79	1.13	0.10 – 2.15	0.52	**0.03**	0.38	-0.05 – 0.81	0.22	0.08
Herbicide	1.98	1.32 – 2.64	0.33	**<0.001**	0.72	-0.17 – 1.62	0.46	0.11	-0.23	-0.61 – 0.14	0.19	0.23
Switchgrass Intercropped	-0.81	-1.45 – -0.18	0.32	**0.01**	-0.69	-1.56 – 0.18	0.44	0.12	0.14	-0.23 – 0.50	0.19	0.47
Random Effects												
σ^2	4.00				7.42				1.31			
τ_{00}	0.04 $_{Stand}$				0.11 $_{Stand}$				0.00 $_{Stand}$			
ICC	0.01				0.01							
N	5 $_{Stand}$				5 $_{Stand}$				5 $_{Stand}$			
Observations	365				365				365			
Marginal R^2 / Conditional R^2	0.27 / 0.27				0.09 / 0.10				0.02 / NA			

Table 2.15 Linear model results comparing species richness for migratory guilds between forest stages within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. Bolded P-values indicate statistical significance (α = 0.05).



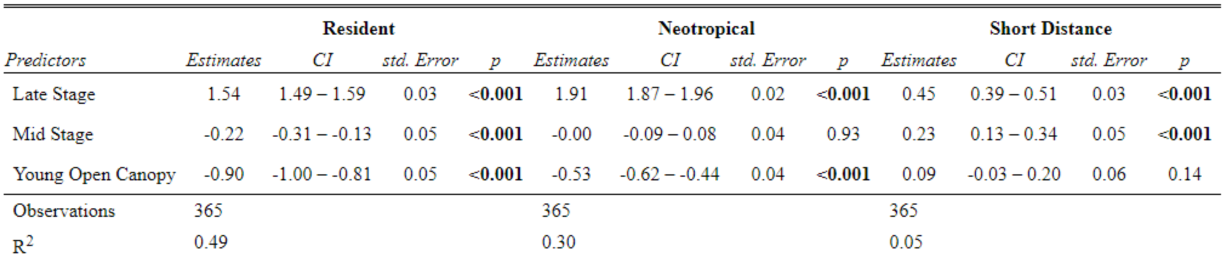
	Resident				**Neotropical**				**Short Distance**			
Predictors	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*
Late Stage	6.58	6.36 – 6.81	0.11	**<0.001**	10.24	9.94 – 10.55	0.16	**<0.001**	2.09	1.94 – 2.25	0.08	**<0.001**
Mid Stage	-1.59	-1.99 – -1.18	0.20	**<0.001**	0.14	-0.41 – 0.70	0.28	0.61	0.62	0.34 – 0.89	0.14	**<0.001**
Young Open Canopy	-4.35	-4.77 – -3.92	0.22	**<0.001**	-4.42	-5.01 – -3.83	0.30	**<0.001**	-0.16	-0.46 – 0.13	0.15	0.28
Observations	365				365				365			
R^2	0.53				0.40				0.06			

Table 2.16 Linear mixed effects model results using treatment as a fixed effect and survey block as a random effect on species diversity for migration guilds within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. Bolded P-values indicate statistical significance (α = 0.05).



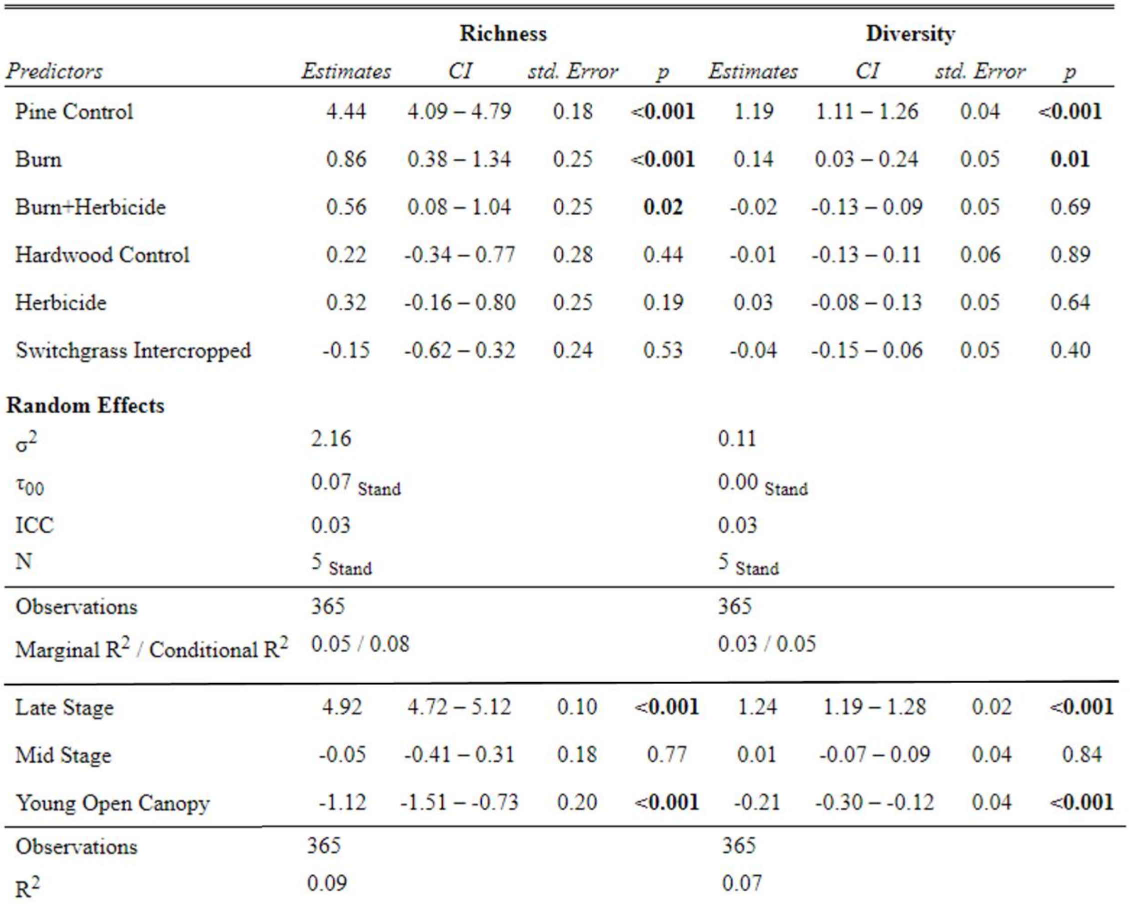
	Resident				**Neotropical**				**Short Distance**			
Predictors	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*
Pine Control	1.17	1.09 – 1.26	0.04	**<0.001**	1.75	1.67 – 1.82	0.04	**<0.001**	0.57	0.49 – 0.64	0.04	**<0.001**
Burn	0.34	0.19 – 0.49	0.08	**<0.001**	0.27	0.15 – 0.39	0.06	**<0.001**	-0.15	-0.29 – -0.01	0.07	**0.04**
Burn+Herbicide	0.40	0.25 – 0.55	0.08	**<0.001**	0.07	-0.05 – 0.19	0.06	0.26	-0.07	-0.21 – 0.07	0.07	0.30
Hardwood Control	0.04	-0.13 – 0.21	0.09	0.66	0.14	-0.00 – 0.28	0.07	0.05	0.05	-0.11 – 0.21	0.08	0.51
Herbicide	0.39	0.24 – 0.54	0.08	**<0.001**	0.10	-0.03 – 0.22	0.06	0.13	-0.19	-0.33 – -0.05	0.07	**0.01**
Switchgrass Intercropped	-0.20	-0.34 – -0.05	0.07	**0.01**	-0.13	-0.25 – -0.01	0.06	**0.04**	0.05	-0.09 – 0.19	0.07	0.47
Random Effects												
σ^2	0.20				0.14				0.18			
τ_{00}	0.00_{Stand}				0.00_{Stand}				0.00_{Stand}			
ICC	0.00				0.01							
N	5_{Stand}				5_{Stand}				5_{Stand}			
Observations	365				365				365			
Marginal R^2 / Conditional R^2	0.19 / 0.19				0.09 / 0.10				0.04 / NA			

Table 2.17 Linear model results comparing diversity indices for migratory guilds between forest stages within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. Bolded P-values indicate statistical significance (α = 0.05).



	Resident				**Neotropical**				**Short Distance**			
Predictors	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*
Late Stage	1.54	1.49 – 1.59	0.03	**<0.001**	1.91	1.87 – 1.96	0.02	**<0.001**	0.45	0.39 – 0.51	0.03	**<0.001**
Mid Stage	-0.22	-0.31 – -0.13	0.05	**<0.001**	-0.00	-0.09 – 0.08	0.04	0.93	0.23	0.13 – 0.34	0.05	**<0.001**
Young Open Canopy	-0.90	-1.00 – -0.81	0.05	**<0.001**	-0.53	-0.62 – -0.44	0.04	**<0.001**	0.09	-0.03 – 0.20	0.06	0.14
Observations	365				365				365			
R^2	0.49				0.30				0.05			

Table 2.18 Linear mixed effects model results comparing species richness and diversity indices for species of conservation concern, using treatment as a fixed effect and survey block as a random effect, and linear model results comparing richness and diversity indices for species of conservation concern between forest stages within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. Bolded P-values indicate statistical significance (α = 0.05).



	Richness				**Diversity**			
Predictors	*Estimates*	*CI*	*std. Error*	*p*	*Estimates*	*CI*	*std. Error*	*p*
Pine Control	4.44	4.09 – 4.79	0.18	**<0.001**	1.19	1.11 – 1.26	0.04	**<0.001**
Burn	0.86	0.38 – 1.34	0.25	**<0.001**	0.14	0.03 – 0.24	0.05	**0.01**
Burn+Herbicide	0.56	0.08 – 1.04	0.25	**0.02**	-0.02	-0.13 – 0.09	0.05	0.69
Hardwood Control	0.22	-0.34 – 0.77	0.28	0.44	-0.01	-0.13 – 0.11	0.06	0.89
Herbicide	0.32	-0.16 – 0.80	0.25	0.19	0.03	-0.08 – 0.13	0.05	0.64
Switchgrass Intercropped	-0.15	-0.62 – 0.32	0.24	0.53	-0.04	-0.15 – 0.06	0.05	0.40
Random Effects								
σ^2	2.16				0.11			
τ_{00}	0.07 $_{\text{Stand}}$				0.00 $_{\text{Stand}}$			
ICC	0.03				0.03			
N	5 $_{\text{Stand}}$				5 $_{\text{Stand}}$			
Observations	365				365			
Marginal R^2 / Conditional R^2	0.05 / 0.08				0.03 / 0.05			
Late Stage	4.92	4.72 – 5.12	0.10	**<0.001**	1.24	1.19 – 1.28	0.02	**<0.001**
Mid Stage	-0.05	-0.41 – 0.31	0.18	0.77	0.01	-0.07 – 0.09	0.04	0.84
Young Open Canopy	-1.12	-1.51 – -0.73	0.20	**<0.001**	-0.21	-0.30 – -0.12	0.04	**<0.001**
Observations	365				365			
R^2	0.09				0.07			

Table 2.19 Models within 2 AIC of the top model for predicting abundance for four avian species within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi from 1999 – 2008 using dynamic (recruitment-survival) N-mixture abundance models. For each model, covariates are listed with parameter estimates (original log [gamma] and logit [omega and p] scale) and 95% confidence (UCI, UCI) intervals in parentheses. Site parameters include percent cover of canopy, graminoid, forbs, shrub, and vine, and the detection parameter included is Julian date (jdate). Observer parameter coefficients not included. Bolded parameters indicate statistical significance (α = 0.05).

		Covariate (estimate [UCI, LCI])			
Species	**Model**	**lambda (Initial Abundance)**	**gamma (Recruitment)**	**omega (Survival)**	**p (Detection)**
Common Yellowthroat	1		**canopy (-0.085 [-0.114, -0.056])** **shrub (-0.034 [-0.068, 0.000])** **vine (-0.022 [-0.038, -0.005])**	**forbs (0.045 [0.000, 0.090])** shrub (-0.059 [-0.127, 0.009])	jdate (-0.072 [-0.185, 0.041])
	2		**canopy (-0.081 [-0.109, -0.053])** **shrub (-0.053 [-0.089, -0.018])** **vine (-0.015 [-0.027, -0.003])**	**forbs (0.072 [0.032, 0.112])**	jdate (-0.0727[-0.190, 0.036])
Kentucky Warbler	1		**graminoid (-0.080 [-0.108, -0.051])** **shrub (0.039 [0.023, 0.275])**		**jdate (0.187 [0.100, 0.275])**
	2		forbs (-0.011 [-0.026, 0.005]) **graminoid (-0.072 [-0.100, -0.045])** **shrub (0.033 [0.015, 0.050])**		**jdate (0.189 [0.101, 0.276])**
	3		forbs (-0.011 [-0.026, 0.004]) **graminoid (-0.078 [-0.112, -0.045])**	**shrub (0.073 [0.023, 0.122])**	**jdate (0.187 [0.100, 0.274])**
	4		**graminoid (-0.091 [-0.125, -0.056])**	**shrub (0.085 [0.034, 0.136])**	**jdate (0.185 [0.098, 0.272])**
	5		**graminoid (-0.087 [-0.122, -0.052])** shrub (0.024 [-0.014, 0.061])	shrub (0.041 [-0.047. 0.130])	**jdate (0.187 [0.100, 0.274])**
	6		forbs (-0.010 [-0.026, 0.005]) **graminoid (-0.078 [-0.111, -0.045])** shrub (0.017 [-0.021, 0.056])	shrub (0.040 [-0.047, 0.127])	**jdate (0.188 [0.101, 0.275])**
White-eyed Vireo	1		**vine (0.017 [0.007, 0.027])** shrub (0.022 [-0.006, 0.051])	shrub (0.007 [-0.029, 0.043])	**jdate (-0.135 [-0.206, -0.065])**
	2		**vine (0.014 [0.005, 0.024])**	shrub (0.027 [0.003, 0.052])	**jdate (-0.135 [-0.205, -0.064])**
	3		**vine (0.017 [0.005, 0.028])** shrub (0.022 [-0.007, 0.052])	forbs (-0.001 [-0.026, 0.0214) shrub (0.007 [-0.030, 0.043])	**jdate (-0.135 [-0.205, -0.064])**

Table 2.19 (continued)

		Covariate (estimate [UCI, LCI])			
Species	**Model**	**lambda (Initial Abundance)**	**gamma (Recruitment)**	**omega (Survival)**	**p (Detection)**
Wood Thrush	1		**graminoid (-0.435 [-0.621, -0.249])**	**graminoid (0.679 [0.091, 1.267])**	**jdate (0.191 [0.083, 0.299])**
	2		**graminoid (-0.485 [-0.714, -0.255])** forbs (0.009 [-0.011, 0.030])	**graminoid (0.680 [0.131, 1.230])**	**jdate (0.191 [0.083, 0.300])**
	3		forbs (0.022 [-0.007, 0.052]) **graminoid (-0.491 [-0.713, -0.269])** vine (0.013 [-0.005, 0.031])	**graminoid (0.701 [0.165, 1.237])**	**jdate (0.189 [0.081, 0.297])**
	4		**graminoid (-0.413 [-0.600, -0.226])** vine (0.005 [-0.008, 0.018])	**graminoid (0.687 [0.090, 1.283])**	**jdate (0.190 [0.082, 0.298])**

Table 2.20 Models within 2 AIC of the top model for predicting abundance for four avian species within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi from 2011 – 2021 using dynamic (recruitment-survival) N-mixture abundance models. For each model, covariates are listed with parameter estimates (original log [gamma] and logit [omega and p] scale) and 95% confidence (UCI, UCI) intervals in parentheses. Site parameters include percent cover of all grass (allgrass), forbs, shrub, and vine, and detection parameters include percent cloud, temperature (temp) and Julian date (jdate). Observer parameter coefficients not included. Bolded parameters indicate statistical significance (α = 0.05).

		Covariate (estimate [UCI, LCI])			
Species	**Model**	**lambda (Initial Abundance)**	**gamma (Recruitment)**	**omega (Survival)**	**p (Detection)**
Common Yellowthroat	1		**allgrass (0.016 [0.010, 0.021])** vine (0.005 [-0.009, 0.020])	**vine (-0.054 [-0.104, -0.004])**	cloud (0.001 [-0.001, 0.003]) **jdate (0.490 [0.401, 0.578])** **temp (0.036 [0.009, 0.063])**
	2		**allgrass (0.016 [0.010, 0.021])** vine (0.005 [-0.009, 0.020])	**vine (-0.054 [-0.104, -0.004])**	**jdate (0.498 [0.401, 0.586])** **temp (0.039 [0.012, 0.065])**
	3		**allgrass (0.016 [0.010, 0.021])**	vine (-0.028 [-0.057, 0.001])	**jdate (0.498 [0.401, 0.585])** **temp (0.042 [0.015, 0.068])**
	4		**allgrass (0.016 [0.010, 0.021])**	vine (-0.029 [-0.058, 0.000])	cloud (0.001 [-0.001, 0.003]) **jdate (0.498 [0.401, 0.577])** **temp (0.039 [0.013, 0.066])**
Kentucky Warbler	1		**allgrass (-0.024 [-0.044, -0.003])** **shrub (0.030 [0.001, 0.059])**		**cloud (-0.008 [-0.014, -0.002])** **jdate (0.367 [0.098, 0.636])**
White-eyed Vireo	1		**forbs (-0.066 [-0.103, -0.029])** **shrub (0.031 [0.019, 0.043])** **vine (0.013 [0.001, 0.025])**	**forbs (0.054 [0.003, 0.104])** **shrub (-0.075 [-0.120, -0.030])**	cloud (0.000 [-0.002, 0.002]) **jdate (0.059 [0.050, 0.067])**
	2		**forbs (-0.061 [-0.100, -0.024])** **shrub (0.031 [0.019, 0.043])** **vine (0.017 [0.002, 0.031])**	**forbs (0.052 [0.001, 0.103])** **shrub (-0.070 [-0.117, -0.024])** vine (-0.017 [-0.057, 0.023])	cloud (0.000 [-0.000, 0.002]) **jdate (0.059 [0.050, 0.067])**
Wood Thrush	1		**vine (-0.612 [-1.065, -0.160])**	shrub (-4.140 [-16.357, 8.086])	temp (-0.155 [-0.339, 0.028]) cloud (0.007 [-0.008, 0.022])
	2		**vine (-0.560 [-1.021, -0.100])**		temp (-0.155 [-0.333, 0.031]) cloud (0.007 [-0.008, 0.022])

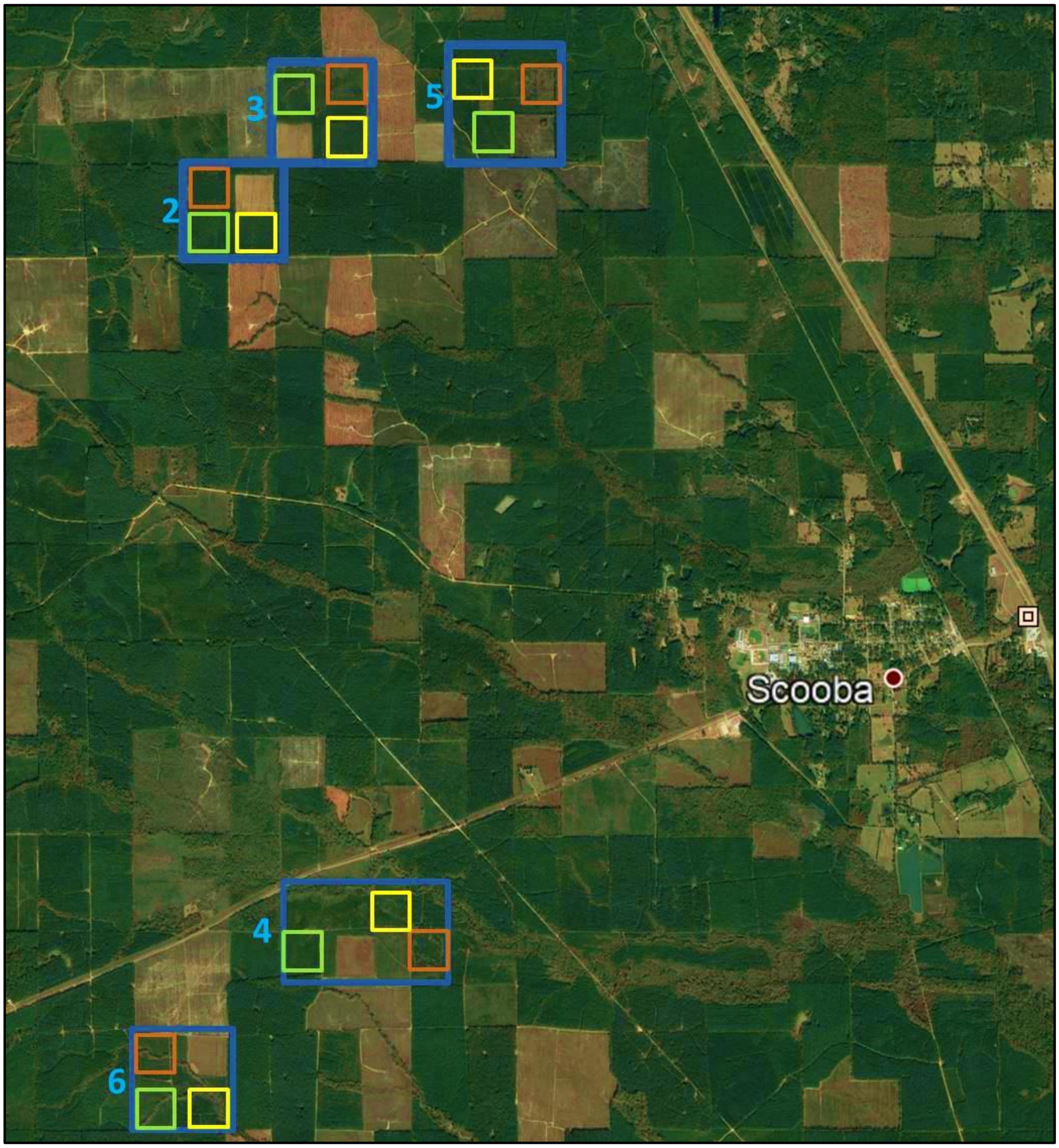


Figure 2.1 Map of study area within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. Blue boxes show boundaries of each study stand. Colored boxes indicate treatment quadrants in 2011: green – Pine Control; brown - Switchgrass Intercropped; and yellow - Hardwood Control (from 2015 on).

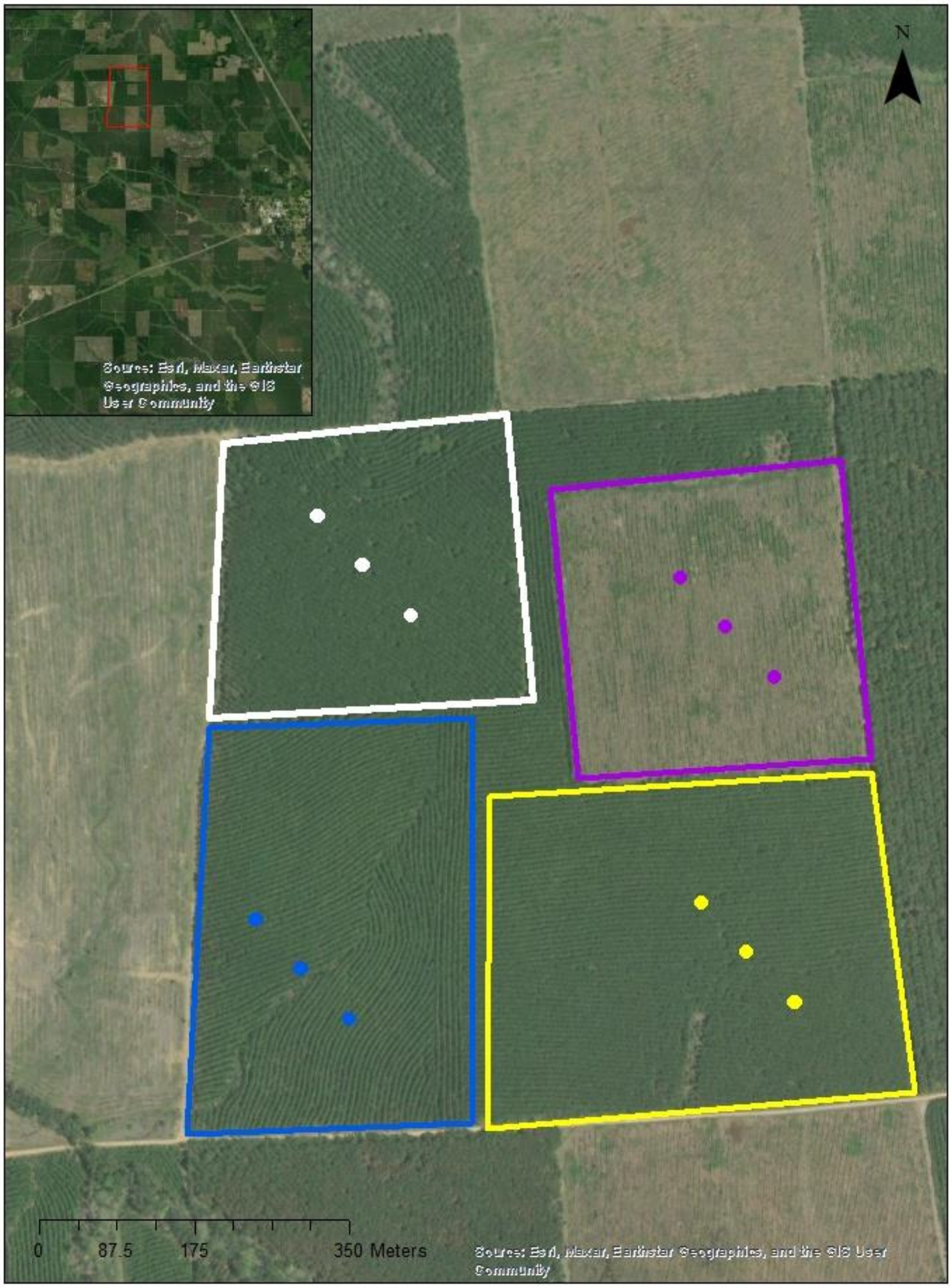


Figure 2.2 Example of point count survey design used during May and June 2011–2021 within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. Points indicate survey point locations while the colored polygons designate the boundaries of each treatment: red - Hardwood Control; yellow - Pine Control; blue - Switchgrass Intercropped; purple - Switchgrass Monoculture (discontinued after 2011 and excluded from study).

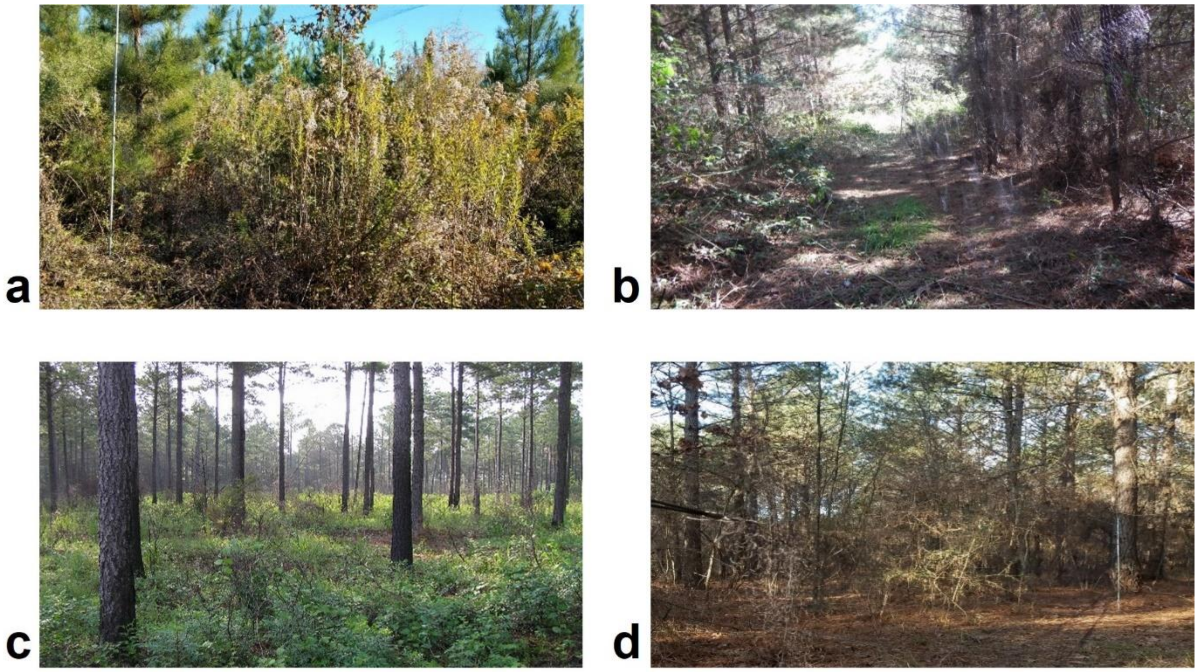


Figure 2.3 Visual examples of forest structure at each forest stage within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi: (a) Young Open Canopy; (b) Mid Unthinned; (c) Mid Thinned; and (d) Late Stage thinned. Mid thinned photo provided by Darren Miller and Angie Larsen-Gray.

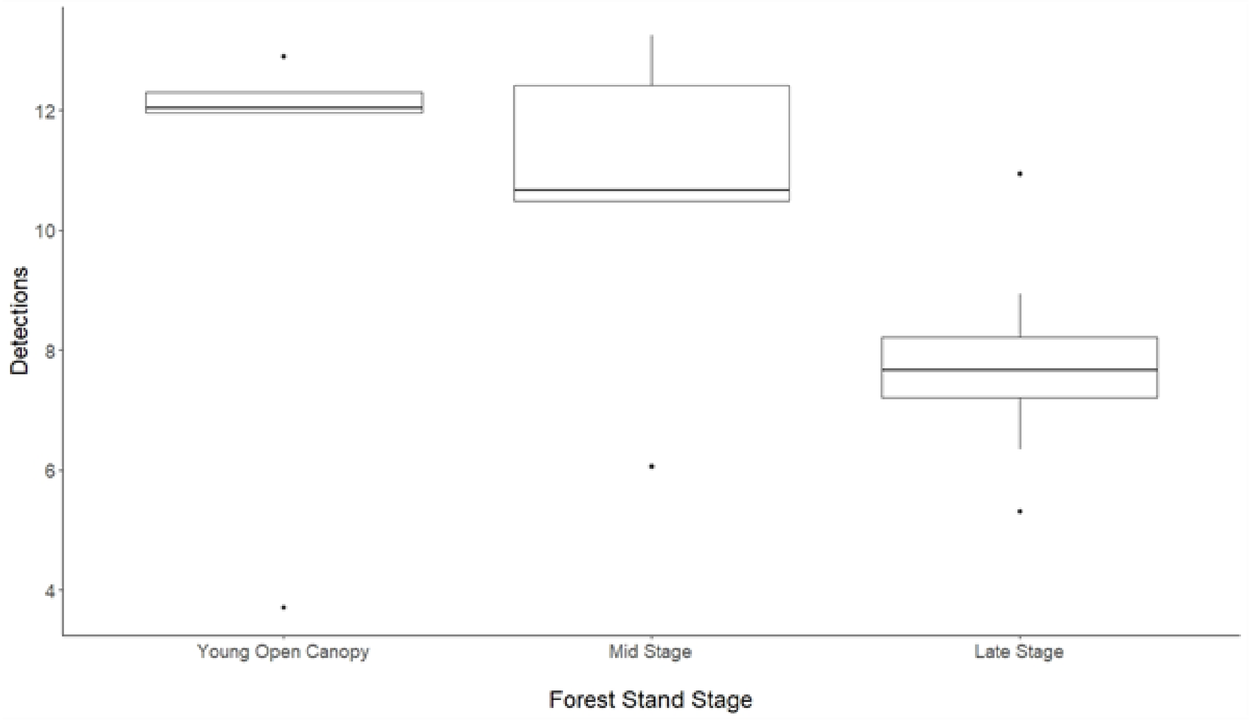


Figure 2.4 Mean number of adjusted detections for all species within 50 meters per survey point in May and June 1999–2008 and 2011–2021 within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi, grouped by forest stand stage.

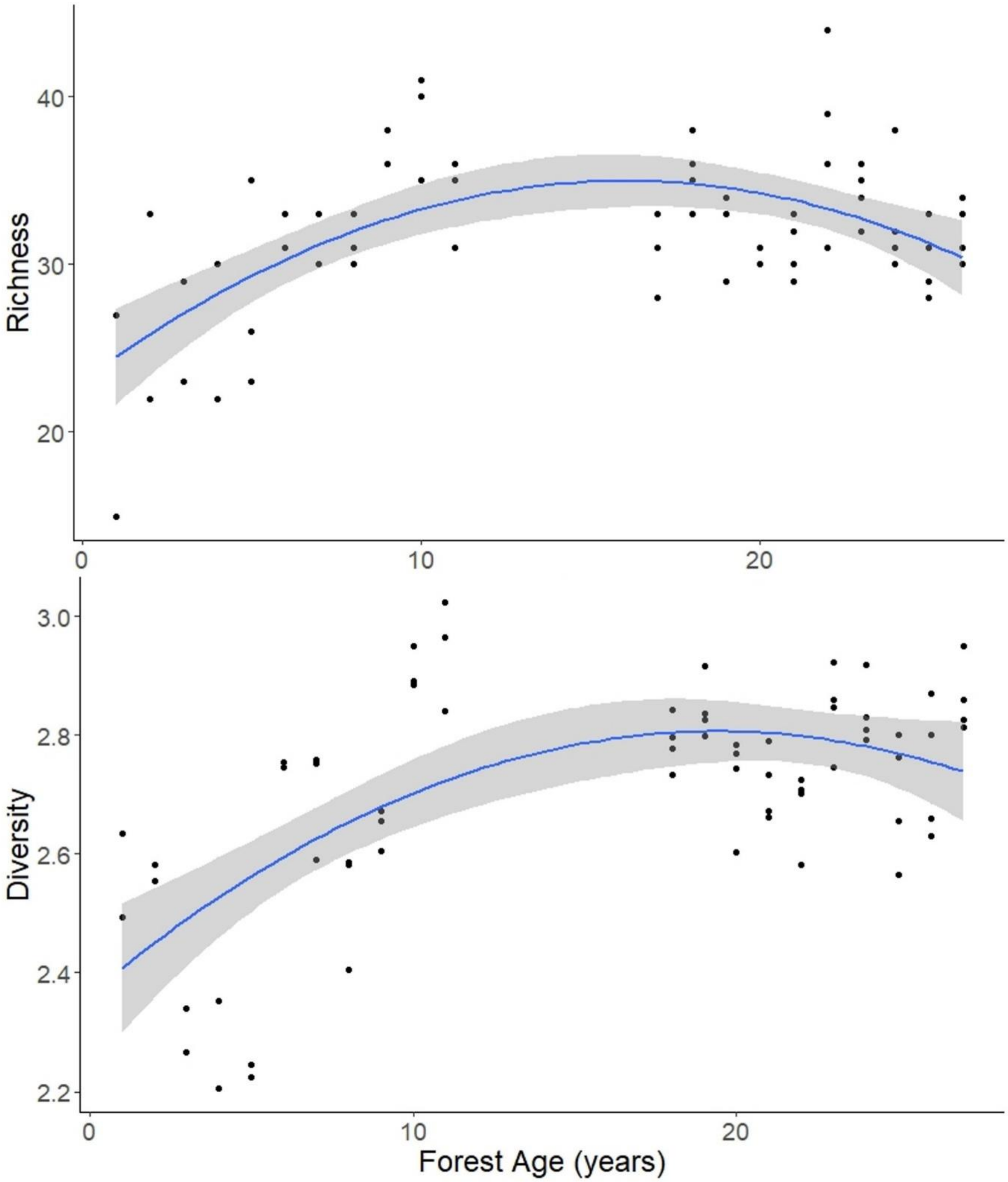


Figure 2.5 Observed and predicted species richness and diversity in response to forest age within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi, including smoothed regression line and 95 % confidence interval. Models include treatment effects.

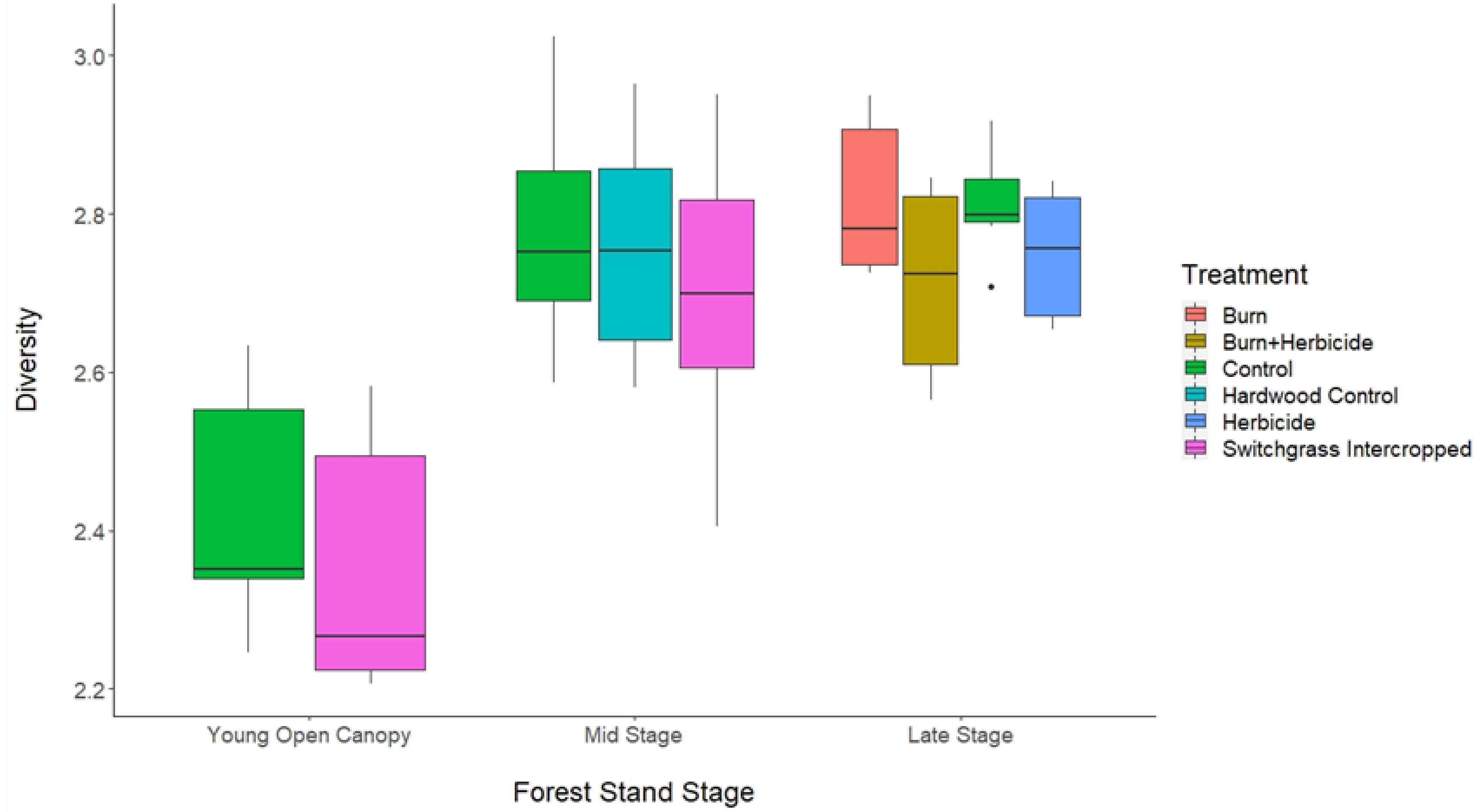


Figure 2.6 Diversity indices grouped by treatment and forest stand stage, from within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi.

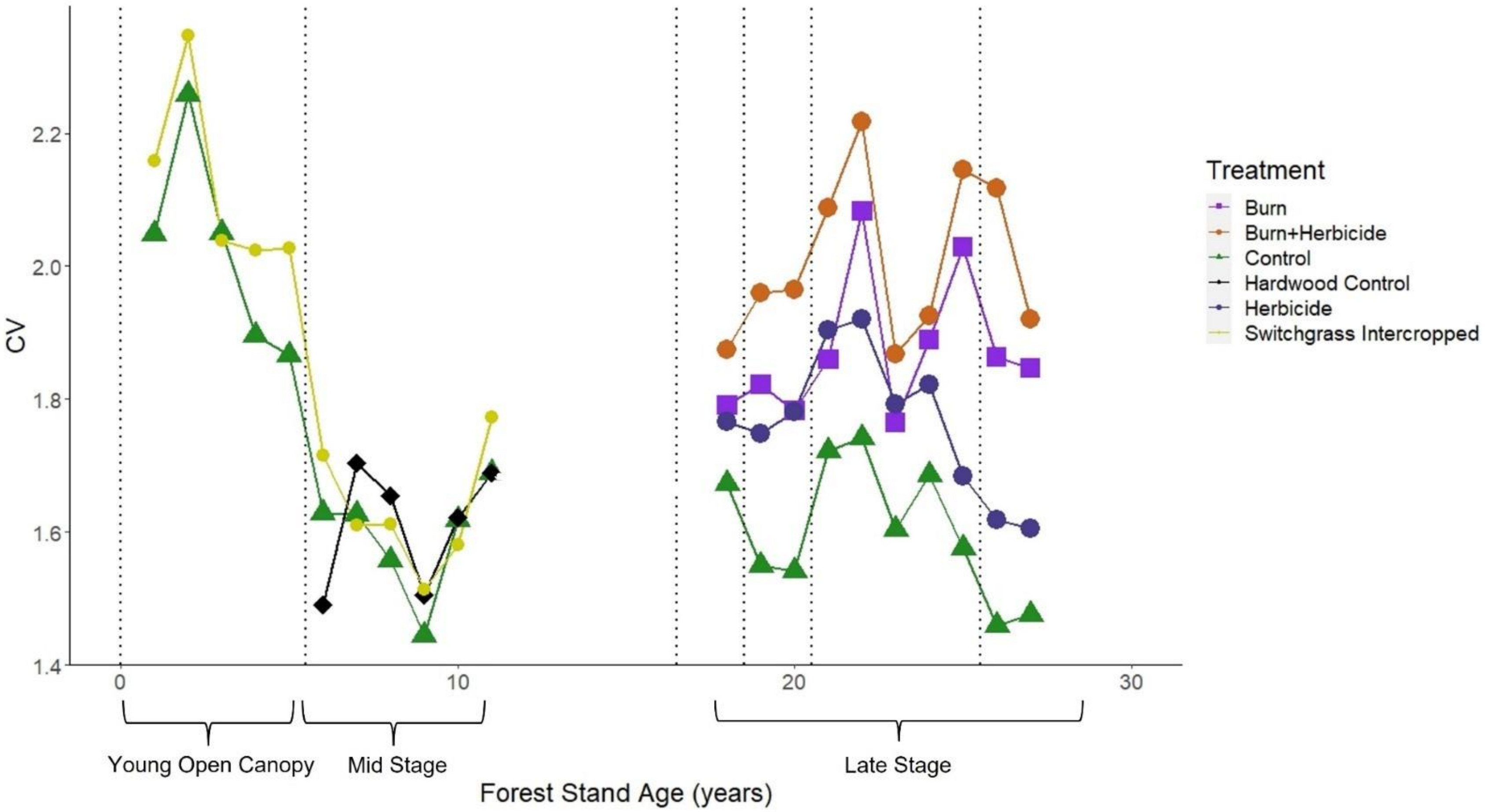


Figure 2.7 Conservation value (CV) indices by forest stand age and treatment, from within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. For ages 17 to 26, forest stand ages were averaged and rounded to the nearest whole number as multiple aged stands were used for each treatment. Dotted vertical lines represent implementation of active management, as described in the methods section of this chapter.

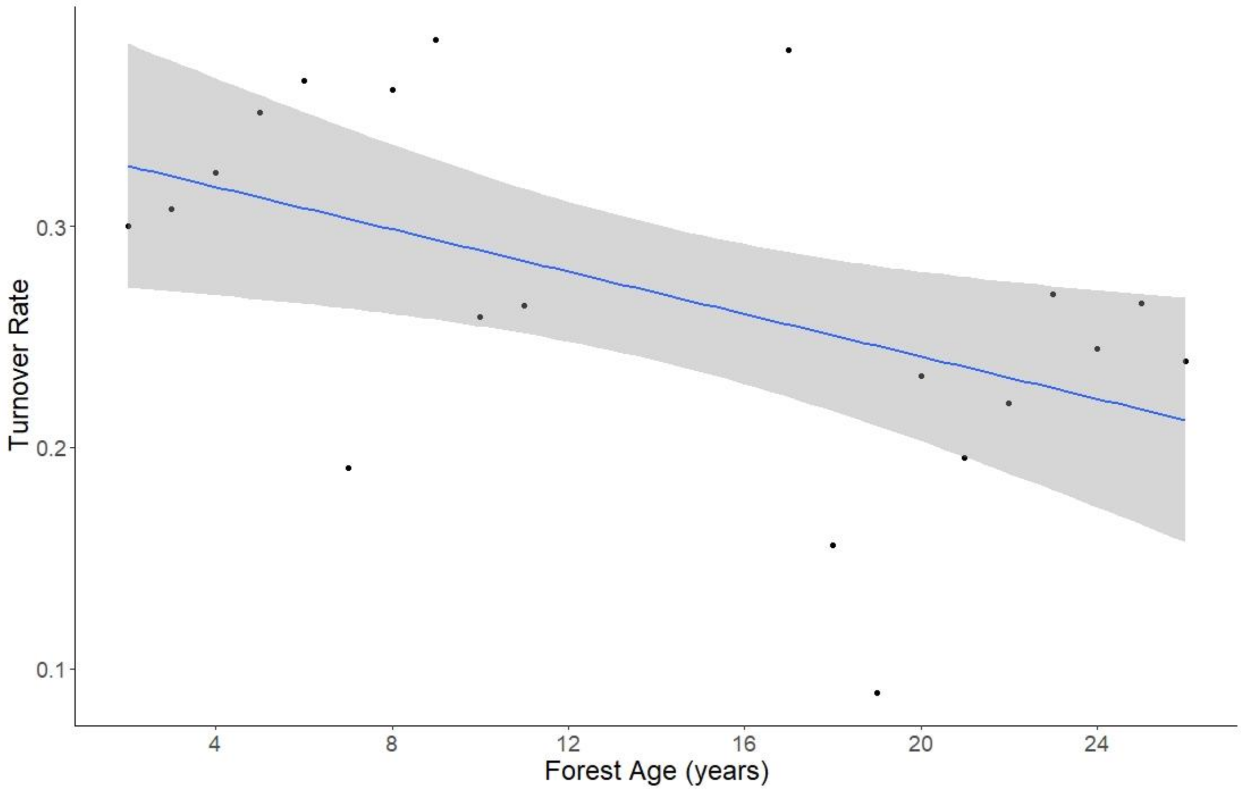


Figure 2.8 Observed and predicted avian turnover rates in response to forest age within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi, including smoothed regression line and 95 % confidence interval.

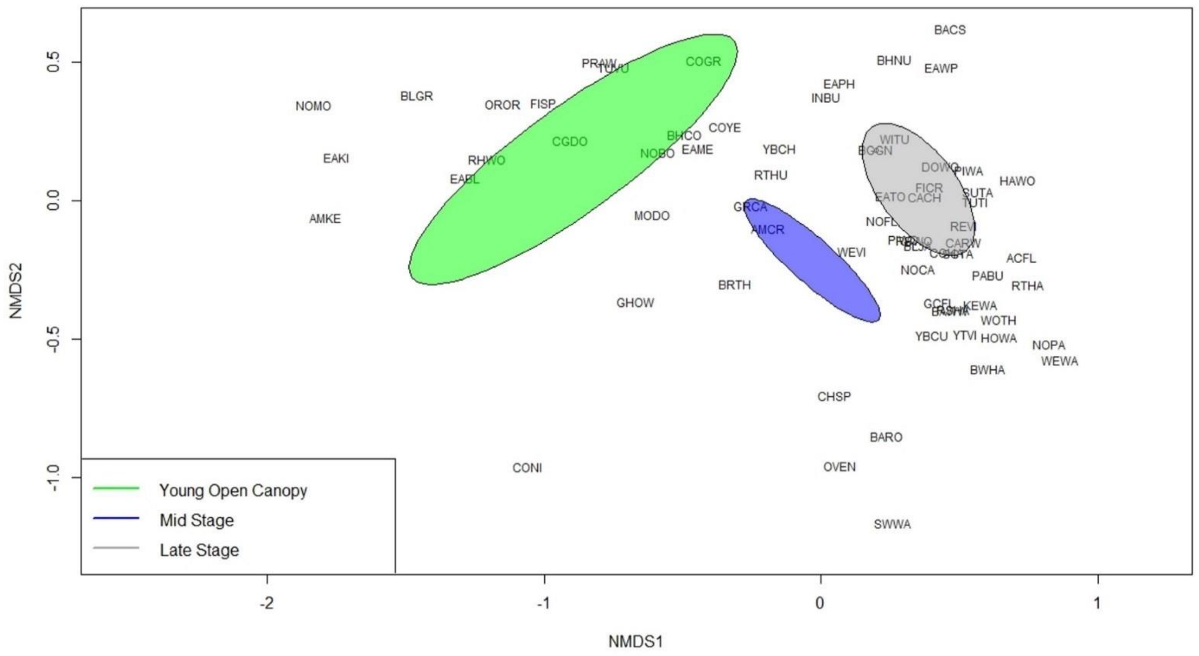


Figure 2.9 Non-metric multidimensional scaling analysis (NMDS), using species detection and forest stage within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. Bird species codes as described in Table 2.2.

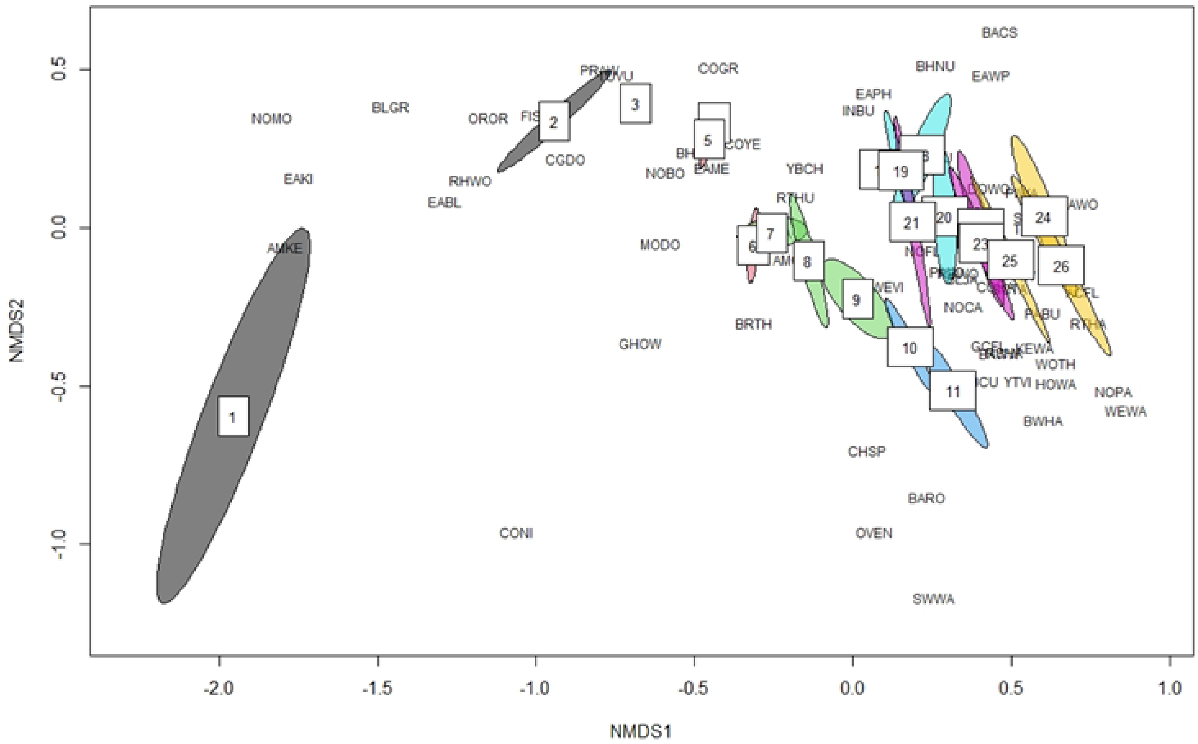


Figure 2.10 Non-metric multidimensional scaling analysis (NMDS), using species detection and forest age within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. Colored ellipses labeled with forest age in years since stand initiation. Bird species codes as described in Table 2.2.

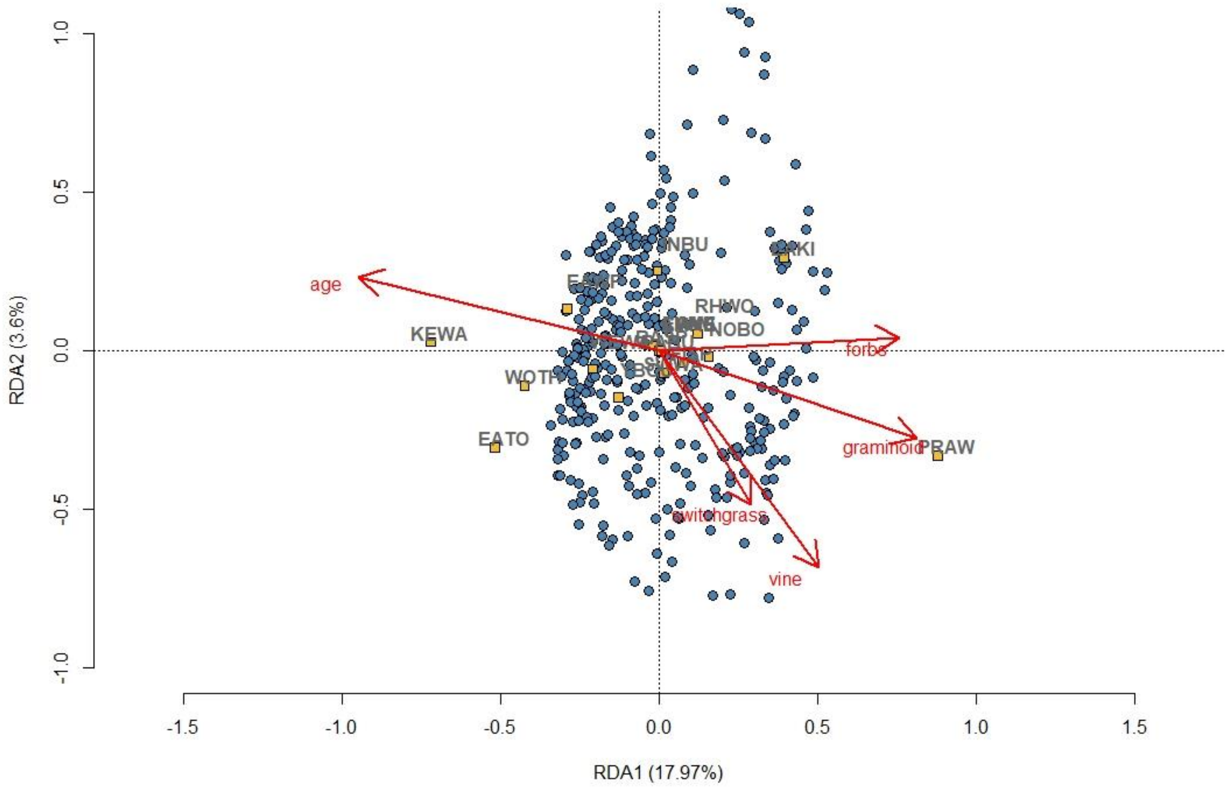


Figure 2.11 First two axes (percentage of variance explained by axis) of ordination by redundancy analysis (optionally principal components analysis) of priority bird species, vegetation structure variables, and forest age within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. Yellow points are bird species (see Table 2.2 for species alpha codes) and blue points represent each row of data, where each row is a treatment within a specific stand in a year. Longer arrows indicate a stronger influence of forest age or vegetation metric on the bird species.

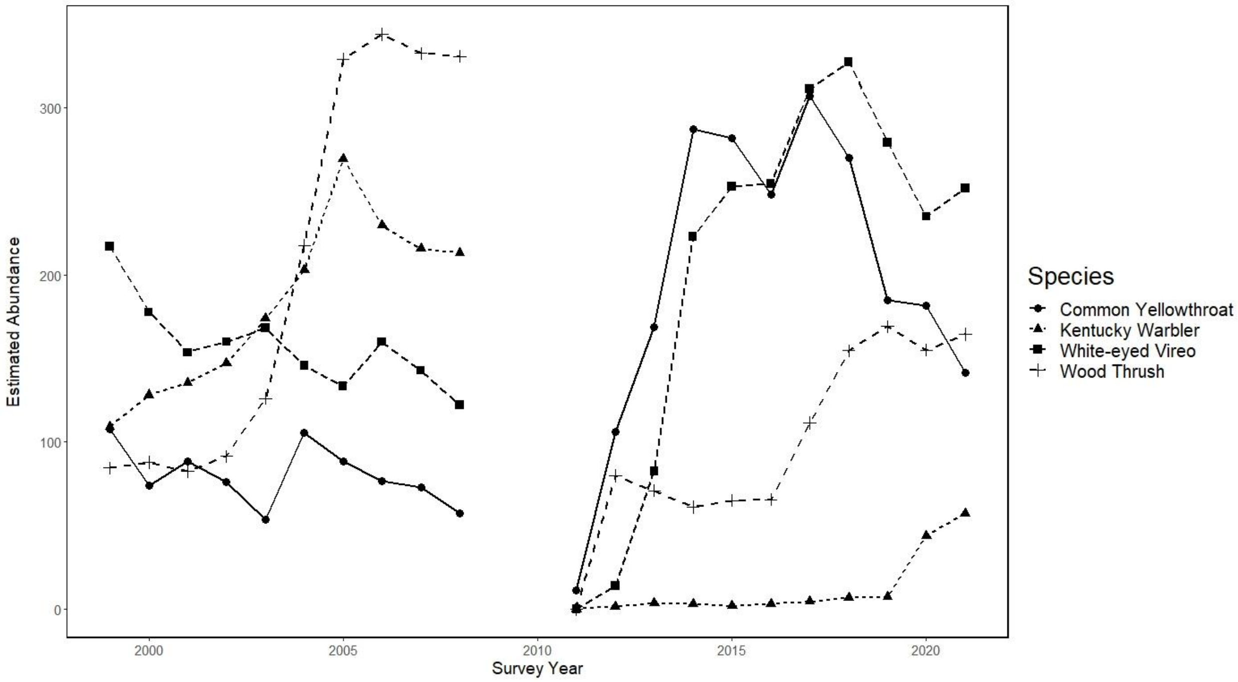


Figure 2.12 Estimated abundances for four avian species by survey year within 5 managed loblolly pine (*Pinus taeda*) stands from 1999–2008 and 2011–2021 in east-central Mississippi.

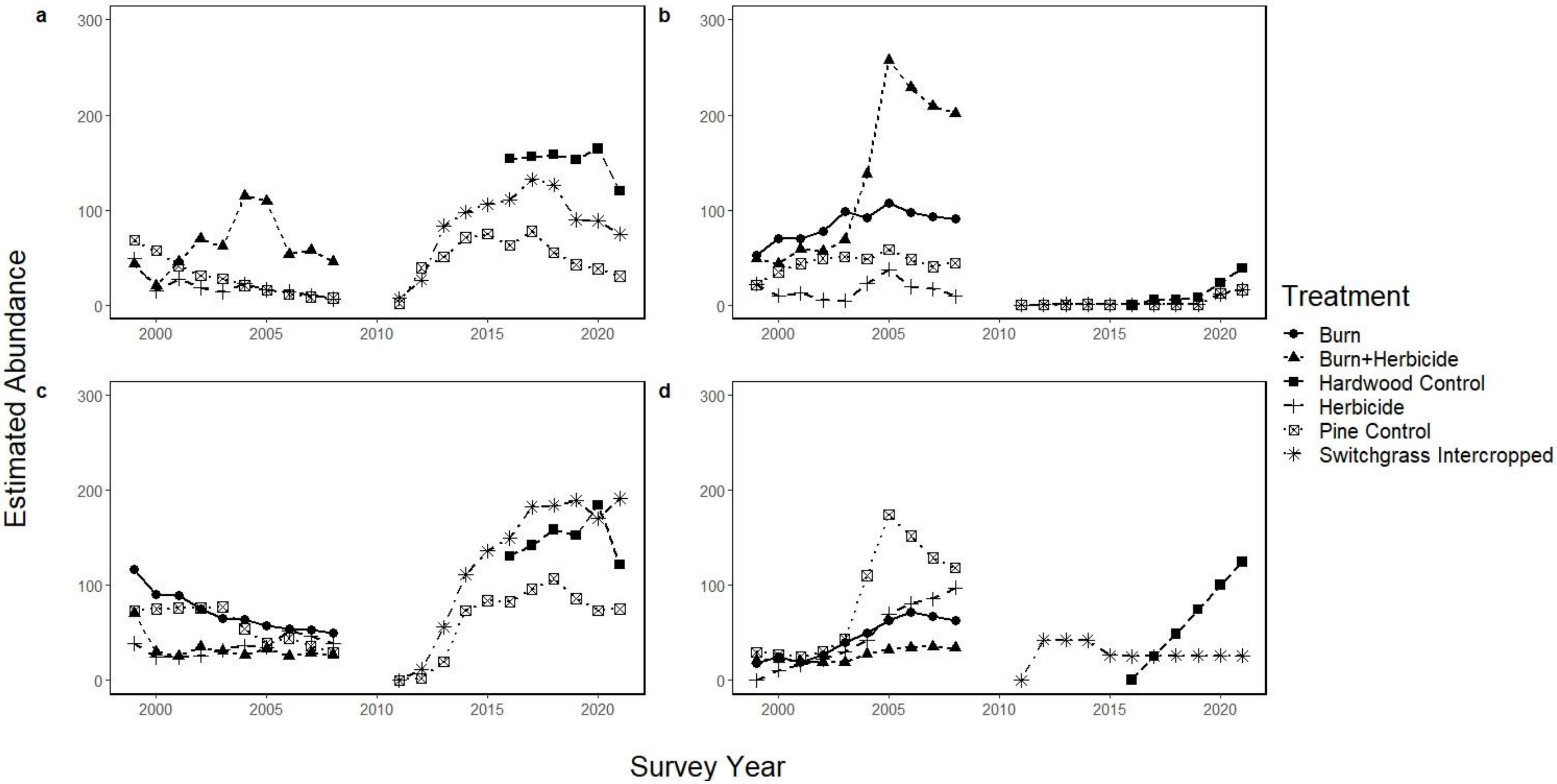


Figure 2.13 Estimated abundances for (a) Common Yellowthroat [*Geothlypis trichas*], (b) Kentucky Warbler [*Geothlypis formosa*], (c) White-eyed Vireo [*Vireo griseus*], and (d) Wood Thrush [*Hylocichla mustelina*] by treatment within each survey year from within 5 managed loblolly pine (*Pinus taeda*) stands, May – June 1999–2008 and 2011–2021 in east-central Mississippi.

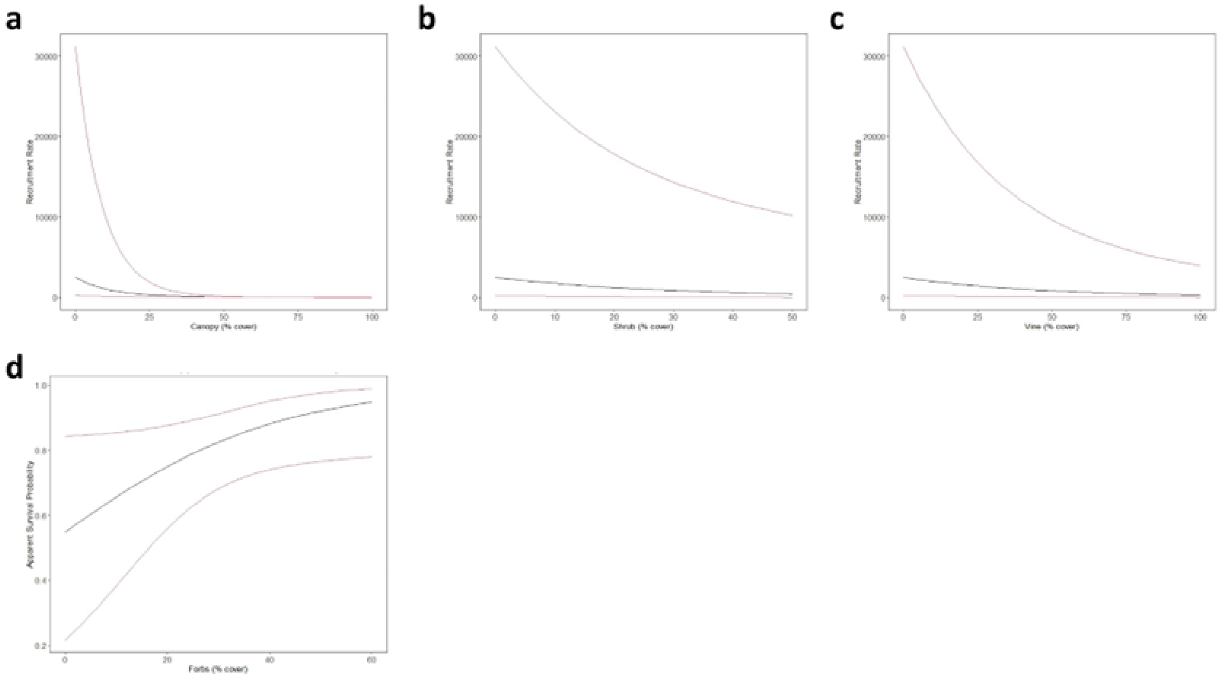


Figure 2.14 Vegetation and observation parameters significantly influencing recruitment (a, b, and c) and apparent survival (d) within dynamic N-mixture abundance models for Common Yellowthroat (*Geothlypis trichas*) within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi from 1999–2008. Red lines represent 95% confidence intervals.

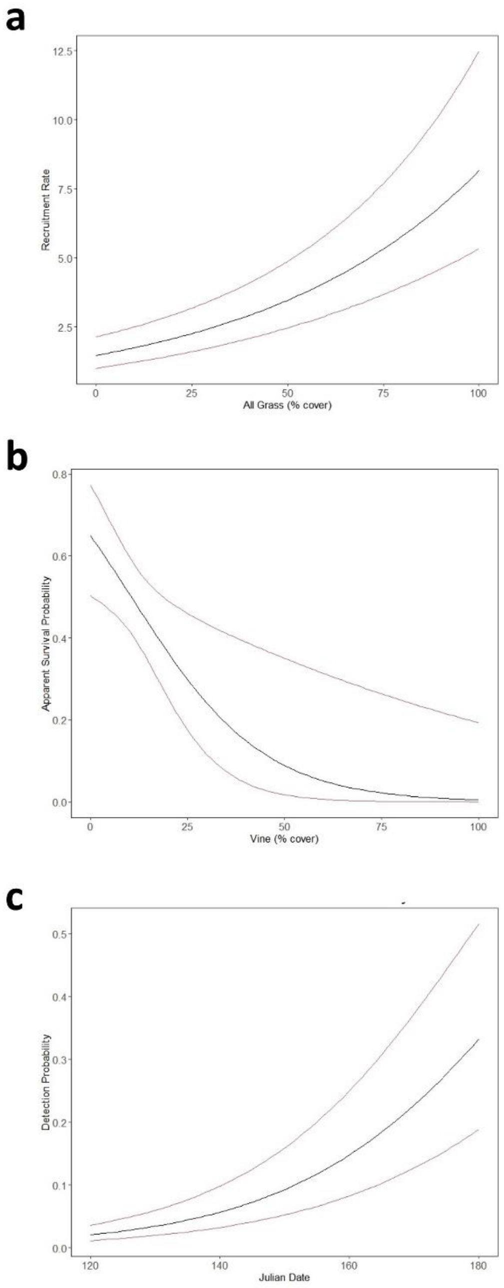


Figure 2.15 Vegetation and observation parameters significantly influencing recruitment (a), survival probability (b) and detection probability (c) within dynamic N-mixture abundance models for Common Yellowthroat [*Geothlypis trichas*] within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi from 2011–2021. Red lines represent 95% confidence intervals.

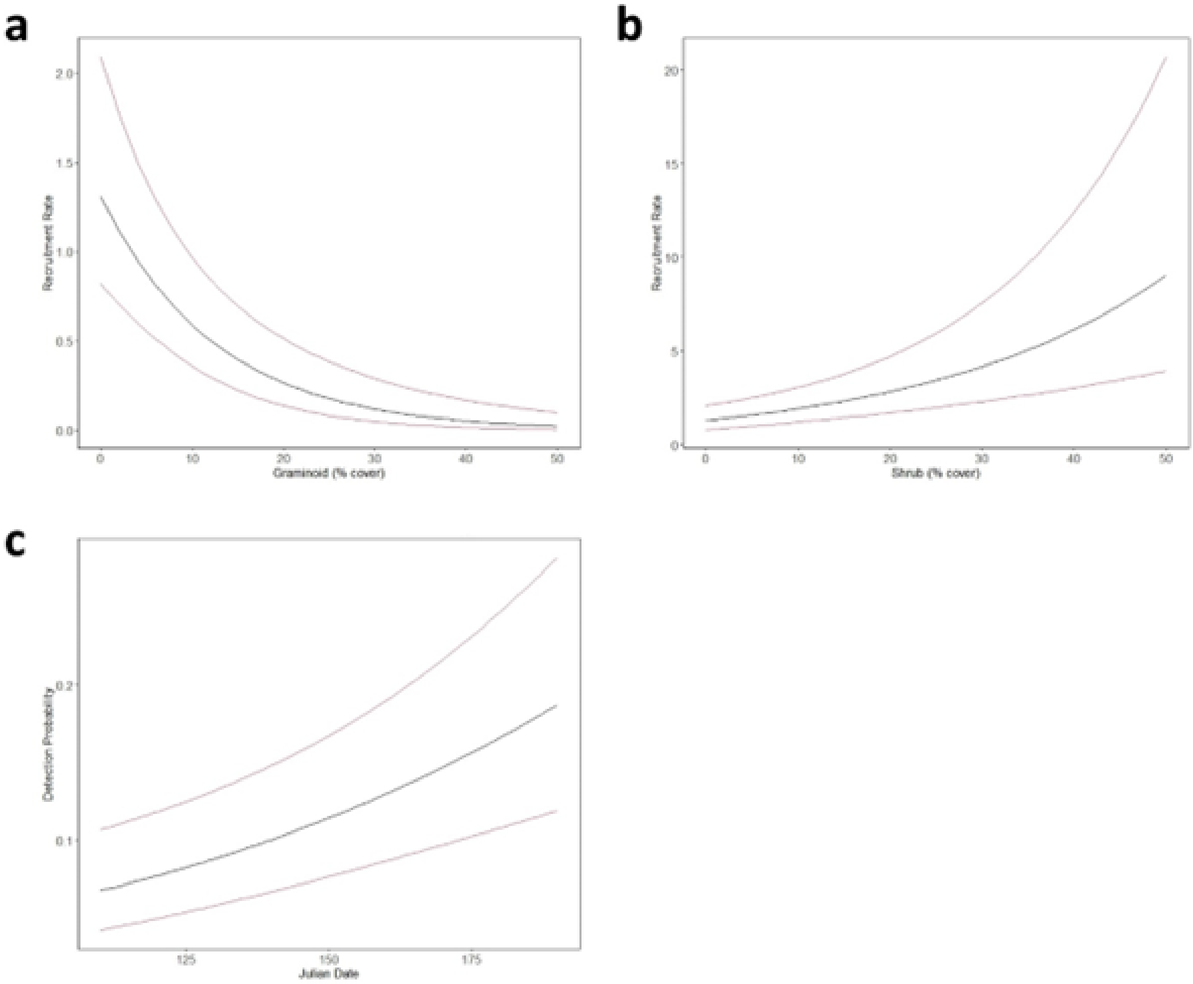


Figure 2.16 Vegetation and observation parameters significantly influencing recruitment (a and b) and detection probability (c) within dynamic N-mixture abundance models for Kentucky Warbler [*Geothlypis formosa*] within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi from 1999–2008. Red lines represent 95% confidence intervals.

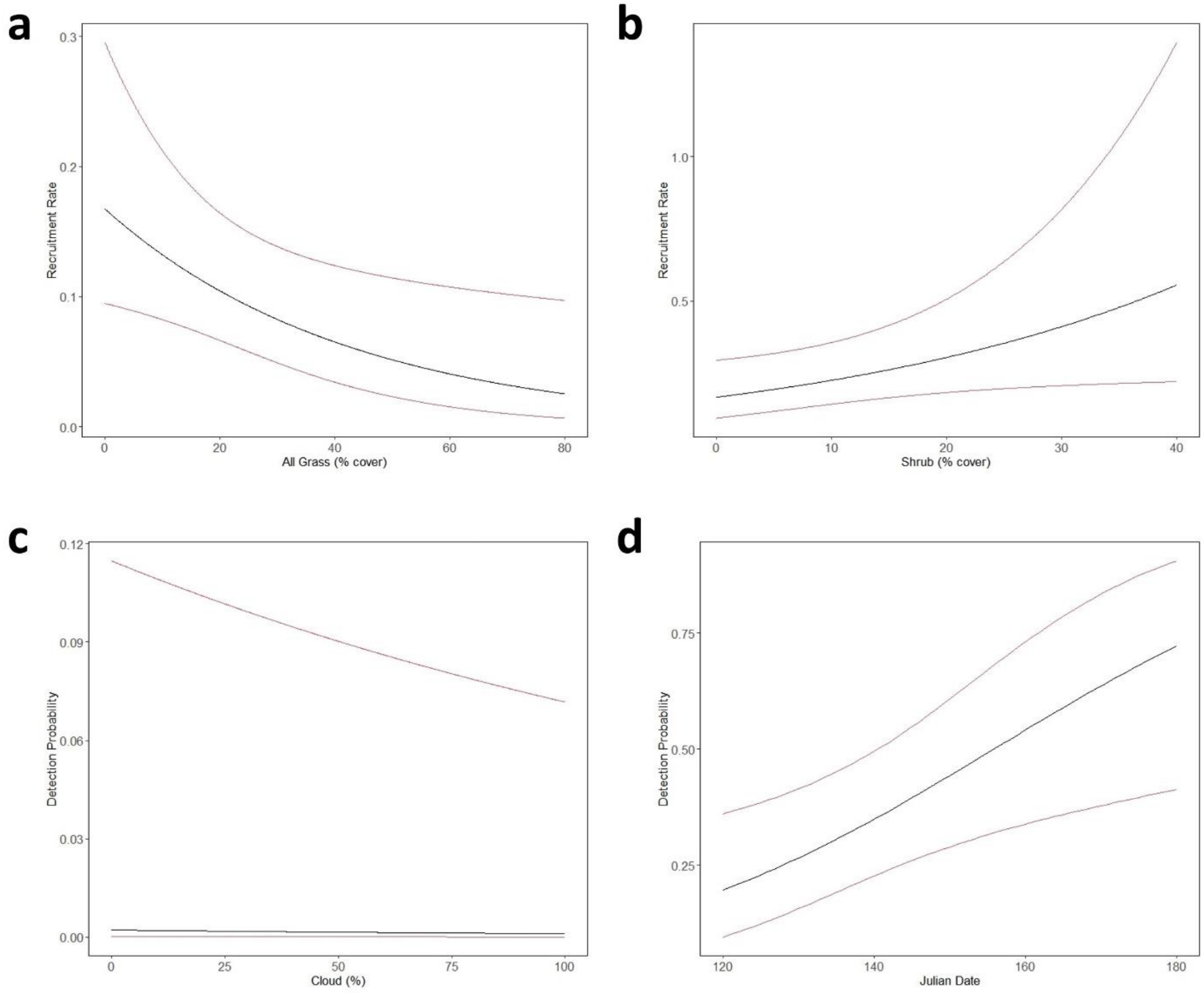


Figure 2.17 Vegetation and observation parameters significantly influencing recruitment (a and b) and detection probability (c and d) within dynamic N-mixture abundance models for Kentucky Warbler [*Geothlypis formosa*] within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi from 2011–2021. Red lines represent 95% confidence intervals.

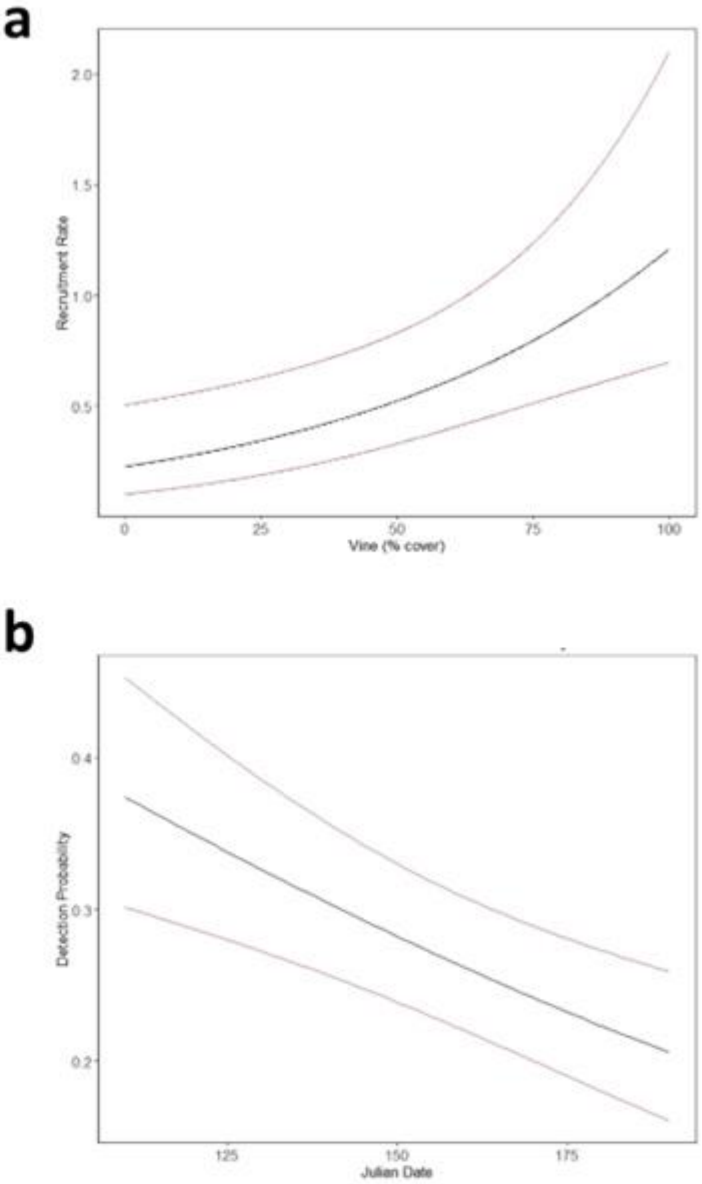


Figure 2.18 Vegetation and observation parameters significantly influencing recruitment (a) and detection probability (b) within dynamic N-mixture abundance models for White-eyed Vireo [*Vireo griseus*] within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi from 1999–2008. Red lines represent 95% confidence intervals.

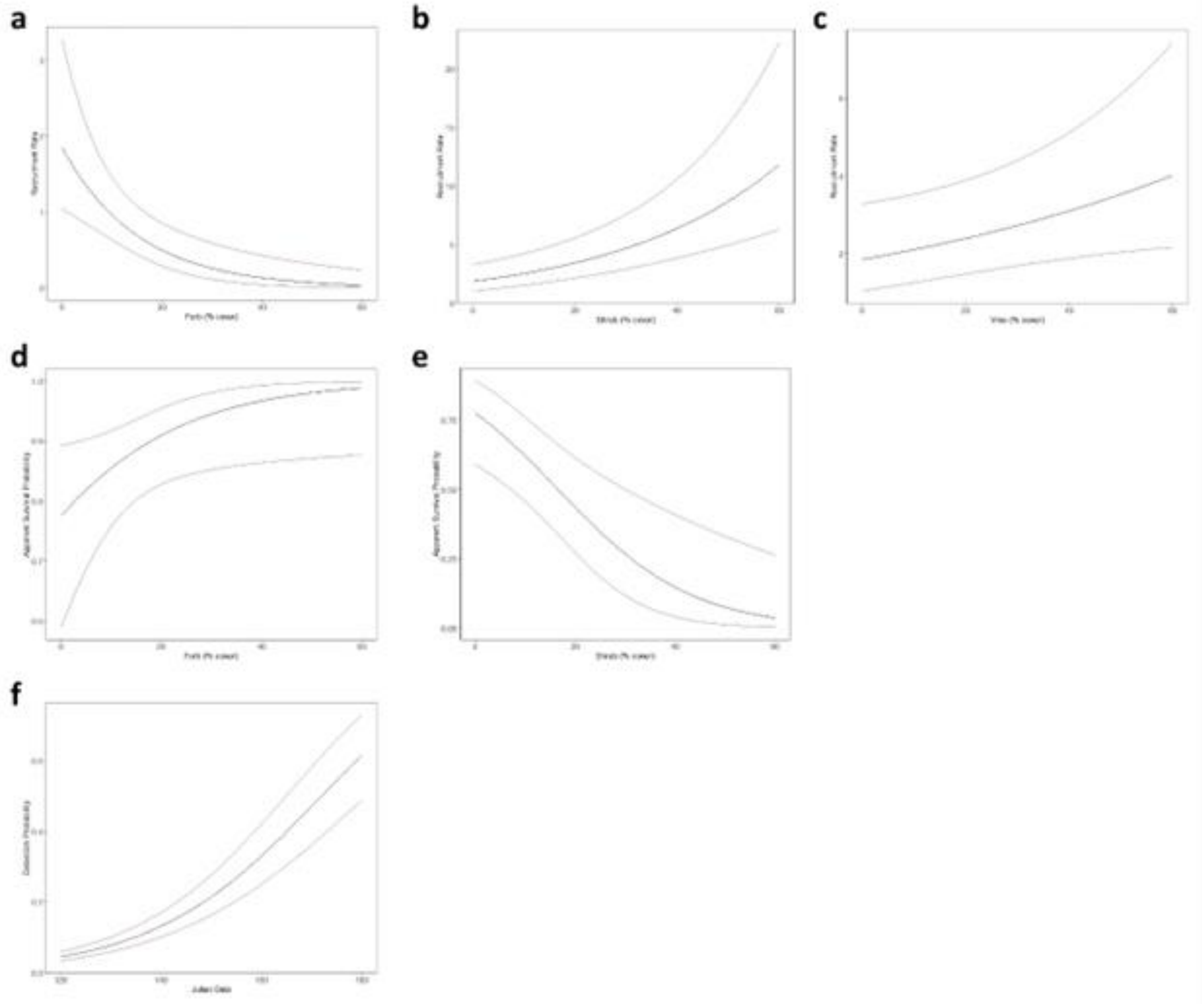


Figure 2.19 Vegetation and observation parameters significantly influencing recruitment (a, b, and c), apparent survival (d and e) and detection probability (f) within dynamic N-mixture abundance models for White-eyed Vireo [*Vireo griseus*] within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi from 2011–2021. Red lines represent 95% confidence intervals.

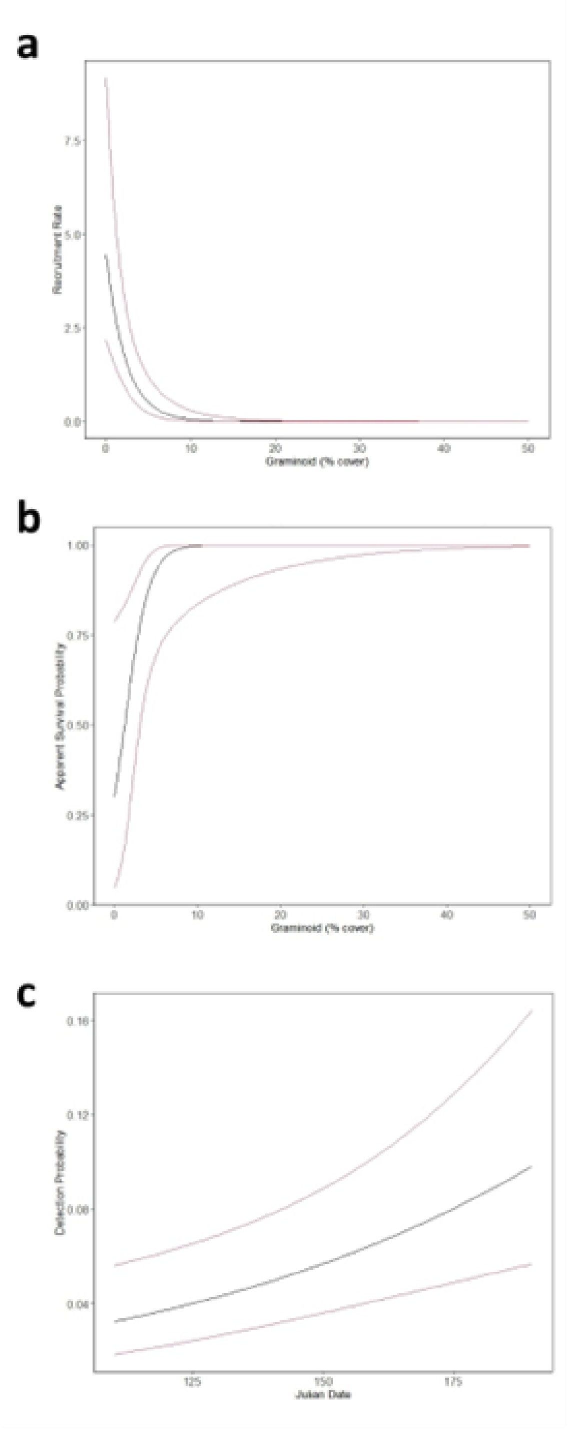


Figure 2.20 Vegetation and observation parameters significantly influencing recruitment (a), apparent survival (b), and detection probability (c) within dynamic N-mixture abundance models for Wood Thrush [*Hylocichla mustelina*] within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi from 1999–2008. Red lines represent 95% confidence intervals.

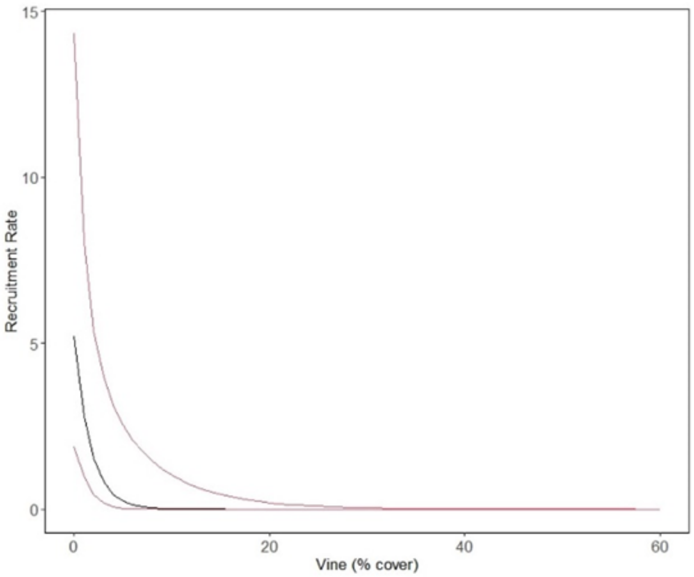


Figure 2.21 Vegetation and observation parameters significantly influencing recruitment within dynamic N-mixture abundance models for Wood Thrush [*Hylocichla mustelina*] within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi from 2011–2021. Red lines represent 95% confidence intervals.

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CHAPTER III

INFLUENCE OF PREVIOUS SWITCHGRASS INTERCROPPING AND HARDWOOD CONTROL WITHIN A MANAGED PINE FOREST ON AVIAN ABUNDANCE AND DIVERSITY

Availability of young, open forest conditions across the U.S. can be considered vitally important in supporting abundances of breeding bird species known to regularly use early successional conditions (Schlossberg and King 2009). With over 80 million hectares of forests suitable for timber production and large increases in managed pine systems since the 1950s, the southeastern U.S. contains large mosaics of forested patches ranging in age and structure (Miller et al. 2009, Hartsell and Connor 2013, U.S. Forest Service 2017). Intensive forest management activities in many of these forests that alter vegetation structure and composition have the potential to significantly affect individual species, and consequently the avian communities, within a particular forest patch and the larger landscape (Guynn et al. 2004, Miller and Miller 2004).

Vegetation structure and composition within working forests is manipulated through management practices (Gundersen et al. 1998, Thysell and Carey 2001). One such practice, planting switchgrass (*Panicum virgatum*) between pine rows, is a method of generating a biofuel feedstock within a short-rotation managed forest (Blazier et al. 2012, Loman et al. 2014, Fuller-Morris et al. 2020). Biofuel and biomass production generally requires substantial amounts of land, highlighting the conundrum facing many land managers as transitioning from forested land to solely biofuel production leads to large emission increases (Cai et al. 2011, Broch et al. 2013). But planting understory vegetation high in lignin and cellulose, such as grasses, creates an opportunity for annual harvesting of a biofuel crop while maintaining merchantable timber products later in the forest rotation (Riffell et al. 2012). Cultivating switchgrass on marginal lands may prove to be more useful in the long term when considering availability and costs of agricultural lands used for bioenergy and food crops (Swinton et al. 2017, Jiang et al. 2019, Fan et al. 2020, Martinez-Feria and Basso 2020). This design can still promote landscape heterogeneity and provide cover for many wildlife species (Hartman et al. 2011).

Tradeoffs may exist when considering implementing biofuel crops within forests. Whereas vegetation structure, plant species diversity, and timing of management and harvest have a clear influence on the occurrence of wildlife species, the ecosystem value of a biofuel crop largely depends on nutrient uptake of the crop, growth and productivity, rate of carbon sequestration, and consequences of replacing other cover types (Fargione et al. 2009). With a positive correlation between biodiversity and ecosystem services, potential biological concessions need to be considered when using a biofuel crop that even temporarily decreases wildlife abundance and ecological processes (Flynn et al. 2009, Duncan et al. 2015, Greene 2016, Brockerhoff et al. 2017). Within agroecosystems, local wildlife diversity is generally enhanced by structurally complex landscapes, with large amounts of heterogeneity in the landscape potentially compensating for high intensity management schemes used at the local scale (Tscharntke et al. 2005, Hartman et al. 2011). Not all biofuel crops have similar impacts on wildlife, with research indicating that using switchgrass over other biofuel crops such as corn can increase herbivore biomass and even nutrient carrying capacities within the system (Robertson et al. 2012, Loman et al. 2017).

When considering impacts on wildlife, specifically breeding bird abundances, the implementation of intercropping switchgrass may not provide migrants and residents with the necessary habitat structure required to sustain significant breeding populations (Loman et al. 2014). Understanding the effects of environmental sustainability as related to the production of biofuels and biomass, particularly their implications for conservation effects and biodiversity sustainability, has become recognized and the focus of research efforts (Fletcher, Jr. et al. 2011, Riffell et al. 2012, Loman 2014, Gottlieb et al. 2017). Biofuels and biomass have the potential to influence conservation efforts positively and negatively, specifically for grassland and early successional bird species, with these species being the focus of several previous studies (Roth et al. 2005, Loman 2014, Grodsky et al. 2016, Marshall 2016, Loman et al. 2018). Grassland species, in particular, are a high priority for research within managed forests and in studies investigating avian responses to biofuel management, likely due to their susceptibility to land changes and in many cases, decreasing populations (Hartman et al. 2011, Wiens et al. 2011, Werling et al. 2014). Nevertheless, previous studies in general did not address the effects of intercropping within managed forest systems, bird species not classified as grassland species, and for extended periods of time during the life of a pine stand (Murray et al. 2003, Roth et al. 2005, Robertson et al. 2011, Loman 2014).

Compared to intercropping, a more common management practice of chemically applying herbicide, such as imazapyr, to each stand allows for an increase in herbaceous vegetation while limiting hardwood encroachment (Welch et al. 2004). A decrease in competitive hardwoods stimulates an increase in forb production and provides the vegetation structure and composition required by many avian species for food sources and nest site availability (Provencher et al. 2001, Singleton et al. 2013, Iglay et al. 2018). With species potentially selecting for specific vegetation structures within known habitat associations (James 1971), an increase in herbaceous vegetation cover and structure can create more opportunities for nests and can provide more conspicuous perching locations (Petit et al. 1988). Conversely, limiting dense shrub cover, often through herbicide application, can lead to a decrease in vegetation structure. Dense, overlapping vegetation, theoretically created by removing hardwoods and allowing for forb growth, in conjunction with increasing amounts of canopy cover and increasing landscape and vegetation heterogeneity, can function as thermal refugia for many wildlife species (Saunders et al. 1998, Palmer et al. 2021). Conversely, retaining non-commercial hardwoods, particularly during the harvesting process, can maintain forest-associated bird species while attracting early seral species (Wheelhouse et al. 2022).

Previous studies in the same working forest found that intercropping switchgrass influenced bird abundances and community composition in the years following stand establishment (Loman 2014, Marshall 2016), and in the years following hardwood control when used five years after stand initiation (Fuller-Morris 2018). But these studies also highlight the potential for avian community convergence between treatments as vegetation recovered in treated areas, with some variation remaining among specific species (Marshall 2016, Fuller-Morris 2018). Concluded from these studies, further assessment was needed to identify community and species-specific trends within treated stands across time, particularly as active management ceased.

Given the importance of conservation-based management for bird species within forested areas in the southeastern U.S., an area dominated by managed pine forests, my main objective was to evaluate avian species and community level responses to different, previously enacted forest management practices. I aimed to further assess avian richness, diversity, and conservation value within loblolly pine (*Pinus taeda*) stands previously intercropped with switchgrass or with hardwood growth controlled. I examined changes in vegetation structure within treatments by survey year to evaluate the influences of prior management. I further assessed how vegetation characteristics influenced species occupancy across time, and if species abundance and occupancy differed between treatments. Finally, I evaluated how avian community structure changed across time within forest stands managed under different methods.

I anticipated avian richness and diversity to be similar among treatments across time, but different among survey years. I predicted that by year ten, vegetation structure would no longer be different among treatments. I hypothesized that percent shrub cover would increase several years post herbicide application, with decreasing cover percentages of grasses, sedges, and rushes in response to canopy closure. However, I anticipated stands intercropped with switchgrass would have less shrub cover and hardwood prevalence as a response to the previous management. I expected to find fewer differences in avian species abundance between treatments when compared to the previous studies, with species occupancy changing dependent on survey year, vegetation structure and composition, and known species habitat association. Finally, I expected control plots to have more bird species associated with woodlands, as stands would more closely mimic later stage forests.

Methods

Study Area and Experimental Design

See Chapter 2 of this dissertation for a full description of study area and experimental design.

Avian Point Count Surveys

See Chapter 2 of this dissertation and Loman (2014), Greene (2016), and Fuller-Morris (2018) for a full description of avian point count survey methods. Avian point count survey data collected during summer 2018 followed Fuller-Morris (2018) protocols. See Figure 3.1 for point count design within one treatment.

Vegetation Surveys

Vegetation surveys were partially described in Chapter 2 of this dissertation. Vegetation surveys followed a three-fold methodology. Following protocols established in a previous study, I completed 25 vegetation surveys per treatment plot, centered along the same diagonal axis as the avian point count surveys (Fuller-Morris 2018). I centered five survey points around each point count survey location and created two additional centers halfway in between each top-middle and middle-bottom point count location (Figure 3.2). Each vegetation survey point was a distance (determined at random) up to 20 meters from, and a random direction from, each center point.

I estimated overstory canopy cover at each point using a concave spherical hand-held densiometer (Forestry Suppliers, Inc. Model C; Lemmon 1956). I acquired a measurement of percent canopy cover for each of the four cardinal directions around the center of the random survey point. I then averaged these four measurements for one measure of canopy cover per survey point.

To determine ground cover density and growth form composition, I used a one-meter quadrat to be centered on each vegetation survey point. Due to the denseness of the understory in most of the plots, I used a one-meter rope centered on the survey point to create a one-meter diameter sampling circle. I estimated the amount of cover as percentages for the following: vine, forbs, graminoids (not including switchgrass), switchgrass, shrubs, hardwood trees, pine trees, dead vegetation, and bare ground. I recorded all estimates using the Daubenmire scale Table 3.1); this standardized the cover percentages as obtaining accurate estimates is difficult and varies by observer (Daubenmire 1959).

To determine vertical plant structure at each vegetation survey point, I used a Nudds board placed at the center of each point (Nudds 1977). The board was divided into six sections, with alternating colors, and was two meters long. Each section was approximately 0.33 meters, with N1 at the bottom and N6 at the top. I stood 15 meters away from the Nudds board in a predetermined random direction and estimated percent covered of each of the six alternatingly colored blocks. The estimated percent was converted to the Daubenmire scale.

See Loman (2014), Greene (2016), and Fuller-Morris (2018) for vegetation survey methods for 2011–2017. Vegetation data from summer 2018 followed protocols established by Fuller-Morris (2018), with the addition of estimating canopy cover using a spherical densiometer.

Statistical Analysis

All analysis was completed in R version 4.0.3 (R Core Team 2020). I removed documented flyovers and flythroughs, and known females, from all models. I also removed non-breeding species, including Bobolink (*Dolichonyx oryzivorus*), Cedar Waxwing (*Bombycilla cedrorum*), Cerulean Warbler (*Setophaga cerulea*), and Sedge Wren (*Cistothorus stellaris*). For all models, results were considered statistically significant at $\alpha < 0.05$.

2019–2021 Richness, Diversity, and Conservation Value

I calculated richness, Shannon-Weaver diversity indices, and Conservation Value (CV) using the *vegan* package (Oksanen et al. 2020) for each treatment within each year and by year. For both richness and diversity metrics I used raw detection counts from all years so species with too few detections for estimating abundance would qualify for diversity analysis. An alternative approach, to use calculated species abundance estimates, would create unequal comparisons between years due to the number of species with estimable abundances each year and would ignore the benefits of including rare species in diversity analyses. To calculate diversity estimates, I used the *diversity* function in the *vegan* package, with the index set to *shannon* for the Shannon-Weaver diversity index (Oksanen et al. 2020). To determine the CV for each treatment within each year, and by year, I calculated a modified Partners in Flight (PIF) rank using Nuttle's method (2003). I multiplied the number of detections for each species by its corresponding modified PIF rank. I then added all values together and divided by the total number of detections for that specific treatment, and year (Loman 2014, Marshal 2016, Fuller-Morris 2018). I then compared diversity indices and CV for each treatment and year using analysis of variance (ANOVA; R Core Team 2020).

2019–2021 Migration and Habitat Guilds

I used the Breeding Bird Survey database to assign a migratory and habitat guild to each species identified during surveys (Sauer et al. 2017; see Table 3.2). I used multiple ANOVA to test for differences in species diversity and CV within species guilds that share traits, specifically migratory status (resident, short-distance migrant, neotropical migrant) and associated breeding habitat (early successional, grassland, and woodland) between treatments and forest age. Grouping species by common traits allowed me to model community group responses versus the response of individual species, as the response of individual species likely varies year to year while the overall community response may remain relatively stable during certain forest succession stages. I checked for normality in diversity and CV within groups using a Shapiro-Wilk normality test within the *rstatix* package, with $P > 0.05$ indicating the distribution is not significantly different from a normal distribution (Shapiro and Wilk 1965, Kassambara 2021).

2019–2021 Avian Species Density Estimates

Using methods established by Loman (2014), Marshall (2016), and Fuller-Morris (2018), I created abundance estimates for each species detected using the *unmarked* package in R (Fiske and Chandler 2011) to compare estimate densities between the Pine Control and the treatments. *Unmarked* allowed me to consider differences in observer ability and varying survey conditions that may influence detection probabilities each year, and the models accounted for heterogeneous detection probabilities (i.e., birds may have been present that were not detected, and therefore not counted). The *unmarked* package also accounts for varied detection probabilities between treatments and years. I removed rare species, species that fit poorly into detection probability models or that did not have sufficient numbers of detections, from further analysis in order to reduce statistical noise. As such, I only modeled species with at least 40 detections per season.

In the first season, individual birds were only recorded within 50m of each survey point while in the two subsequent years detections were recorded at any distance but truncated to individuals detected within 100m of survey point to reduce the likelihood of inaccurate distance estimates. While previous studies have cut all detections at 50m to prevent possible overlapping detections between survey points, I chose not to cut all detections to 50m for the two years where data existed beyond that distance as many species are able to be detected beyond 50m (Ralph et al. 1995). This prevents removing data that may increase statistical accuracy of the models used for estimating abundance, specifically by improving modeling accuracy of shape parameters in distance-based abundance estimations, and by capturing distance-based detection errors that may have occurred as distance to detection increase (Alldredge et al. 2007). Detections beyond this threshold were discarded because density estimates depend on having a set measure of sample area and distance estimations vary not only by observer but also by species and cover type, decreasing in detection probability as distance increases (Buckland et al. 1993, Farnsworth et al. 2002).

To calculate densities, I fit generalized distance sampling models using the *gdistsamp* function and tested a no covariate model to determine the best key function (exponential, halfnorm, hazard, or uniform), which described the detection function shape based on the documented distance values, via Akaike Information Criterion (AIC; Akaike 1973, Fiske and Chandler 2011). I then determined if a Poisson or negative binomial distribution was most appropriate for each species distribution. Finally, I tested the effect of treatment as an abundance (state) covariate, and cloud and temperature as detection covariates as each was recorded during the five survey replicates per season. For the 2020 models, where there were two different observers throughout the season, I also tested for observer effects as a detection covariate. I set the Pine Control treatment as the reference, to compare the two other techniques (Hardwood Control and Switchgrass Intercropping) to standard management. I used the best fit model as determined by lowest AIC and determined density estimates (number of males per hectare) for each species using the *predict* function in *unmarked* (Fiske and Chandler 2011). If treatment was not included in the best fit model for a species, I only reported total predicted density estimates. I checked model fit using the Freeman-Tukey fit statistic (Freeman and Tukey 1950) for the observed data and compared to expected values generated from 100 simulations (Fiske and

Chandler 2011). I used the *fitstats* function in the *AHMbook* package to calculate the bootstrap P-value as the proportion of expected values greater than the observed value, with a $P > 0.05$ indicating a good-fitting model (R Core Team 2020, Kéry et al. 2021). I considered density estimates between treatments and Pine Control to be significantly different if $P < 0.05$ for estimated β coefficients.

2019–2021 Vegetation Characteristics

I compared understory density and plant cover between treatments and years using ordinal logistic regression models in the *ordinal* package, with treatment as a fixed effect and plot with each treatment as a random effect (Christensen 2019). I compared treatments to Pine Control plots in all models. I created one model for each Nudds board score (N1 as the lowest section [closest to the ground] to N6 as the highest) per treatment and compared treatment effects. I then used the same model for each vegetation category recorded: Forb (herbaceous flowering plants), Hardwood (all hardwood trees), Shrub (small to medium size woody plants), Graminoid (non-switchgrass grasses, sedges, and rushes), and Vine (woody and herbaceous). I did not model effects of treatment on Switchgrass as that specific vegetation only existed in one treatment, with minimal to no detections in the Hardwood Control and Pine Control plots, nor Bare Ground (no vegetation or only pine needles), Dead Plant Material (including of all types of dead vegetation), and Pine (all pine species). For all models, a < 0.05 was considered significant.

2011–2021 Avian Species Occupancy

Using data provided Loman (2014), Marshall (2015), Greene (2016), and Fuller-Morris (2018), I combined 11 years of bird point count and vegetation survey data to model effects of active management after forest stand initiation, as well as effects post management. I only used detections up to 50 m to standardize effort based on detection distance across all years. I fit dynamic (colonization-extinction) hierarchical occupancy models using the *colext* function in *unmarked* (Fiske and Chandler 2011) to estimate detection probability, initial occupancy, and colonization and extinction rates for five species. Each model equation included four components: psi (ψ), formula for initial probability of occupancy, gamma (γ), formula for colonization probability, epsilon (ε), formula for extinction probability, and p formula for detection probability. I modeled three woodland species and two early successional/scrub species: Common Yellowthroat (*Geothlypis trichas*), Kentucky Warbler (*Geothlypis formosa*), Prairie Warbler (*Setophaga discolor*), Swainson's Warbler (*Limnothlypis swainsonii*), and Wood Thrush (*Hylocichla mustelina*).

I used a binomial response of 0 (unoccupied) or 1 (occupied) per survey point for each survey replicate (five per season), by treatment and year, for each species. I used observer, temperature (°C), wind speed (km/hr), cloud cover (percent), and Julian date as detection covariates and site covariates of Shrub, Forbs, Graminoid, Switchgrass, Vine, AllGrass (graminoid and switchgrass values combined) and Canopy Cover (all in percent) as initial occupancy, colonization, and extinction covariates. I scaled temperature and wind speed in all models. I first tested detection effects by creating single detection covariate models, and then tested all possible combinations of site covariates in combination with the chosen detection covariates. I identified competing models to be those with an AIC of 4 or less.

I created a global model with all covariates that had or were close to statistical significance. I assessed possible overdispersion of parameters for the global model by using the MacKenzie and Bailey *mb.gof.test* with 1000 bootstrap iterations (MacKenzie and Bailey 2004). If overdispersion in the global model was found (c-hat > 2), I corrected competing models by weighting models by the overdispersion parameter and used a quasi-likelihood-based information theoretic approach by computing the quasi-likelihood AICc (QAICc, AICc for overdispersed count data; Burnham and Anderson 2002). I then ranked weighted models in a model selection table using the QAICc weighted values (Table 3.3). If overdispersion was not identified in the global model (c-hat < 2), I ranked models by AICc in a separate model selection table (Table 3.4). I included a no covariate (null) model in all model selection tables.

I retrieved yearly and treatment-based occupancy estimates using *ranef*; by summing all columns, I was able to estimate posterior distributions of the random variables by year and treatment, resulting in estimates of occurrence (Fiske and Chandler 2011). I used *confint* to extract 95% confidence intervals from included covariates (R Core Team 2020) and used the projected model occupancy estimates to evaluate yearly occupancy by treatment and across all sites (Fiske and Chandler 2011).

2011–2021 Avian Community Structure

To further describe avian community changes in response to the changing forest age and characteristics over time, I used the *metaMDS* function in the *vegan* package in R to complete Nonmetric Multidimensional Scaling (NMDS) analysis (Oksanen et al. 2020). I identified dissimilarities between survey year and treatment, reduced dimensionality to the first two ordination axes, and plotted model results to visually assess community level trends as forest age increased. I assessed the fit of environmental independent variables (forest age and treatment) in the ordination model using *envfit* in the *vegan* package (Oksanen et al. 2020). I created three models, one including communities from 2019–2021 separated by survey year, one including communities 2019-2021 separated by treatment, and one including communities from 2011–

2021 separated by treatment and forest stage (Young Open Canopy [survey years 1–5] and Mid Unthinned [survey years 6–11]).

Results

2019–2021 Diversity, Richness, and Conservation Value

From 2019 to 2021, I recorded 7377 detections of 61 breeding bird species (Table 3.2). I had 2386 detections in 2019, 3019 in 2020, and 1972 in 2021 (Table 3.5). Species richness was highest in the Hardwood Control plots in 2020 and lowest in the Pine Control plots in 2019 and Switchgrass Intercropped plots in 2021 (Table 3.6). Yearly species richness was highest in 2020 with 52 species. Diversity was highest in the 2020 Switchgrass Intercropped plots (div = 3.04) and lowest in the 2019 Hardwood Control plots (div = 2.61). Total diversity was highest in 2021 (div = 3.06). CV was lowest in the 2019 Pine Control plots (CV = 1.44) and highest in the 2021 Switchgrass Intercropped plots (CV = 1.78), with increasing values in all treatments between 2019 and 2021 (Table 3.6). ANOVA results indicated that from 2019–2021 there were statistically significant differences in diversity and CV as a response to year ($P_{diversity} = 0.02$, $P_{CV} < 0.001$; Table 3.7), with both diversity and CV increasing from 2019 to 2021.

2019–2021 Migration and Habitat Guilds

For migration guilds, there was a significant response of year on the corresponding diversity and CV in short distance ($P_{div} < 0.01$, $P_{CV} = 0.01$) and neotropical ($P_{div} < 0.01$, $P_{CV} < 0.01$) with diversity estimates increasing from 2019 to 2021 and CV decreasing across the same time period (Table 3.7). For habitat guilds, there was a significant CV response to treatment within the early successional/scrub guild ($P = 0.02$) and a significant CV response to year within the woodland guild ($P < 0.01$; Table 3.7). For early successional/scrub species, CV was higher in the

Switchgrass Intercropped than the Pine Control plots across all years. For woodland species, CV increased from 2019 to 2021.

2019–2021 Avian Species Density Estimates

I used 11 species in 2019, 17 species in 2020, and 15 species in 2021 for density (number of males per hectare) analyses based on the previously mentioned conditions. In 2019, density in Hardwood Control and Switchgrass Intercropped plots compared to Pine Control plots were significantly different for two species: Common Yellowthroat (estimate ± SE, Pine Control 1.72 ± 0.54, Switchgrass Intercropped 5.30 ± 1.36) and Eastern Towhee (Pine Control 10.14 ± 8.96, Switchgrass Intercropped 18.68 ± 16.63; Table 3.8). Both species had higher estimated densities in both the Hardwood Control (Hardwood Control 4.18 ± 1.15 and 18.07 ± 16.26, respectively) and Switchgrass Intercropped plots when compared to the Pine Control plots, with the highest estimates in the Switchgrass Intercropped plots. Four additional species had statistically different density estimates in Switchgrass Intercropped plots when compared to Pine Control plots: Brown-head Cowbird (*Molothrus ater;* Pine Control 24.44 ± 38.51, Switchgrass Intercropped 84.46 ± 128.89), Indigo Bunting (Pine Control 1.06 ± 0.42, Switchgrass Intercropped 2.88 ± 0.89), and Yellow-breasted Chat (*Icteria virens;* Pine Control 3.94 ± 0.74, Switchgrass Intercropped 4.49 ± 8.72) had higher estimates while Hooded Warbler (*Setophaga citrina;* Pine Control 9.35 ± 6.03, Switchgrass Intercropped 1.27 ± 0.83) had a lower estimate in the Switchgrass Intercropped versus the Pine Control plots (Table 3.8). In addition, Carolina Wren (*Thryothorus ludovicianus*) and Pine Warbler (*Setophaga pinus*) had statistically higher estimated densities in Hardwood Control plots when compared to Pine Control plots (Pine Control 4.34 ± 2.24, Hardwood Control 10.60 ± 9.82).

In 2020, only Common Yellowthroat and Indigo Bunting had significantly different densities between Pine Control plots and Hardwood Control and Switchgrass Intercropped plots (Table 3.9). Both species had higher density estimates in the treatments than in the Pine Control plots, with Common Yellowthroat having the highest estimated density in the Hardwood Control plots (Pine Control 0.50 ± 0.82, Hardwood Control 4.11 ± 7.28, Switchgrass Intercropped 3.07 ± 5.38) while Indigo Bunting density estimates were highest in the Switchgrass Intercropped plots (Pine Control 0.06 ± 0.04, Hardwood Control 0.39 ± 0.17, Switchgrass Intercropped 0.55 ± 0.20). Two species, Carolina Wren (Pine Control 1.54 ± 1.25, Hardwood Control 2.68 ± 2.21) and Eastern Towhee (Pine Control 1.31 ± 0.23, Hardwood Control 2.04 ± 0.31), had significantly higher densities in Hardwood Control plots than Controls (Table 3.9). Finally, Yellow-breasted Chat had significantly higher densities in Switchgrass Intercropped plots than in Pine Control plots (Pine Control 1.57 ± 0.25, Switchgrass Intercropped 2.74 ± 0.34; Table 3.9). Density estimates for Wood Thrush were just beyond significance ($P = 0.06$) for Switchgrass Intercropped plots.

In 2021, four species had significantly different densities in Switchgrass Intercropped plots than Pine Control plots (Table 3.10). Great Crested Flycatcher (*Myiarchus crinitus;* Pine Control 21.55 ± 24.25, Switchgrass Intercropped 6.74 ± 8.10) and Hooded Warbler (Pine Control 22.35 ± 17.59, Switchgrass Intercropped 12.17 ± 9.83) density estimates were lower in the Switchgrass Intercropped plots when compared to Pine Control plots, while White-eyed Vireo (Pine Control 2.49 ± 0.48, Switchgrass Intercropped 5.91 ± 0.81) and Yellow-breasted Chat (Pine Control 0.96 ± 0.21, Switchgrass Intercropped 2.21 ± 0.35) had significantly higher density estimates in the Switchgrass Intercropped plots (Table 3.10). Only one species, Yellow-billed

Cuckoo, had significantly lower density estimates in Hardwood Control plots (Pine Control 2.18 ± 2.56, Hardwood Control 0.55 ± 0.65).

The remaining modeled species did not have significantly different density estimates between the Pine Control and treatments. This included, in 2019, Northern Cardinal (*Cardinalis cardinalis*), White-eyed Vireo (*Vireo griseus*), and Wood Thrush, in 2020, Black-and-white Warbler (*Mniotilta varia*), Blue Jay (*Cyanocitta cristata*), Carolina Chickadee (*Poecile carolinensis*), Hooded Warbler, Kentucky Warbler, Mourning Dove (*Zenaida macroura*), Northern Bobwhite, Northern Cardinal, Red-bellied Woodpecker, White-eyed Vireo, and Yellow-billed Cuckoo, and in 2021, Black-and-white Warbler, Blue Jay, Carolina Wren, Common Yellowthroat, Eastern Towhee, Indigo Bunting, Kentucky Warbler, Mourning Dove, Northern Cardinal, and Wood Thrush (Table 3.11).

Early successional/scrub breeding birds had varying responses to the treatments. Common Yellowthroat had higher estimated densities in Switchgrass Intercropped plots for 2019 and 2020, as did Indigo Bunting. Eastern Towhee exhibited higher densities in both Hardwood Pine Control and Switchgrass Intercropped plots for all three years. However, Hooded Warbler had lower density estimates in both Hardwood Control and Switchgrass Intercropped plots compared to Pine Control plots. Similarly, woodland species had mixed responses to treatments across all years. Black-and-white Warbler density estimates in Pine Control plots in 2020 and 2021 and Kentucky Warbler estimates in Hardwood Control plots were not statistically significant from Pine Control plots. Yellow-billed Cuckoo initially had similar estimates in all plots, but in 2021 had significantly higher estimates in Pine Control plots.

Generalist species, specifically White-eyed Vireo and Yellow-breasted Chat, had relatively similar density estimates across all treatments across time, though Yellow-breasted

Chat had significantly higher estimates in Switchgrass Intercropped treatments. Brown-headed Cowbird was only modeled one year (2019; Table 3.8), and while models showed adequate fit, estimates were likely skewed by flocking behavior.

While all models demonstrated adequate fit, by Freeman-Tukey test statistic ($P > 0.05$) and c-hat between 0.7 and 1.1, several species for at least one year had standard errors larger than predicted density estimates, indicating less precise density estimates (e.g., Blue Jay, Brown-headed Cowbird, Carolina Chickadee, Common Yellowthroat, Northern Cardinal, Northern Bobwhite, and Red-bellied Woodpecker).

2019–2021 Vegetation Characteristics

Nudds board readings for 2019 indicated that understory vegetation in Switchgrass Intercropped plots and Hardwood Control plots were significantly less dense at multiple vertical Nudds board categories compared to Pine Control plots (Figure 3.3). For Switchgrass Intercropped plots there was a significant difference ($P < 0.05$) at N1 (0–0.33 m), N4 (1–1.33 m), N5 (1.33–1.67 m), and N6 (1.67–2 m) and at N4 (1–1.33 m), N5 (1.33–1.67 m), and N6 (1.67–2 m) for Hardwood Control. In 2020, Hardwood Control plots were less dense ($P < 0.05$) at N3 (0.67–1 m), N4 (1–1.33 m), N5 (1.33–1.67 m), and N6 (1.67–2 m), compared to Pine Control plots, as were Switchgrass Intercropped at N4 (1–1.33 m; Figure 3.4). In 2021, Hardwood Control plots were less dense ($P < 0.05$) than Pine Control plots at N1 (0–0.33 m), N2 (0.33–0.67 m), N5 (1.33–1.67 m), and N6 (1.67–2 m), as were Switchgrass Intercropped plots at N4 (1–1.33 m) and N6 (1.67–2 m; Figure 3.5). This indicates that, overall, vertical structure and vegetation density was less dense in actively managed plots when compared to Pine Control plots that had not been managed since stand initiation.

For modeled vegetation cover, there was significantly less Vine coverage in Hardwood Control and Switchgrass Intercropped plots compared to Pine Control plots for all three years ($P < 0.05$; Figures 3.6, 3.7, and 3.8). Contrastingly, there was significantly more Shrub cover in both Hardwood Control and Switchgrass Intercropped plots, compared to Pine Control plots, for all years ($P < 0.05$). For Forb cover, there was a statistically equal amount between treatments and Pine Control in 2019 and 2021, with less cover in the Switchgrass Intercropped plots compared to Pine Control plots in 2020. Similarly, there was a statistically significant decrease in amount of hardwood cover in Hardwood Control plots compared to Pine Control plots in 2020, but no difference between treatment and Pine Control plots in 2019 and 2021. Graminoid cover was statistically less in Switchgrass Intercropped and Hardwood Control plots compared to Pine Control plots in 2019, but no difference was documented in 2020 and 2021.

2011–2021 Avian Species Occupancy

Global models for Common Yellowthroat, Prairie Warbler, Swainson’s Warbler and Wood Thrush showed characteristics of overdispersion and were corrected using the dispersion parameter (c-hat$_{\text{commonyellowthroat}}$ = 3.08, c-hat$_{\text{prairiewarbler}}$ = 3.36, c-hat$_{\text{swainson's warbler}}$ = 2.46, c-hat$_{\text{woodthrush}}$ = 2.30; Table 3.3). After weighting competing models with the dispersion parameter, no vegetation covariate was included in the top model for Common Yellowthroat, though percent cover of vines ($P = 0.07$, $\beta = 0.129$, SE = 0.070) was close to significantly influencing the extinction probability (Table 3.12). Competing models for estimating Kentucky Warbler occupancy indicated that the amount of vine ($P = 0.26$, $\beta = 0.081$, SE = 0.071) and graminoid ($P = 0.68$, $\beta = -0.019$, SE = 0.047) cover, while included, did not influence extinction probabilities. Percent cover of graminoid ($P = 0.77$, $\beta = 0.031$, SE = 0.104) also did not influence the colonization probability, with no significant vegetation covariates in the competing models (Table 3.12).

While shrub cover was the only vegetation covariate included in the top model for Prairie Warbler (Table 3.12), it produced no effect on the extinction probability ($P = 0.39$, $\beta = -0.075$, SE = 0.088). Percent cover of forbs was present in both the colonization ($P = 0.08$, $\beta = 0.034$, SE = 0.019) and extinction ($P < 0.01$, $\beta = -0.126$, SE = 0.047) formulas in the top model for Swainson's Warbler (Table 3.12). Increasing forb cover resulted in decreasing extinction estimations and had no significant effect on colonization (Figure 3.9). Finally, for Wood Thrush, increasing percentage of graminoid cover ($P = 0.13$, $\beta = -0.055$, SE = 0.036) had no effect on extinction probability in the top occupancy models, while increasing vine cover ($P = 0.15$, $\beta = -0.068$, SE = 0.048) had no effect on colonization rates.

Julian date had a considerable influence on detection probabilities for Common Yellowthroat ($P < 0.01$, $\beta = -0.147$, SE = 0.055; Figure 3.10A), Kentucky Warbler ($P < 0.001$, $\beta = 0.058$, SE = 0.000; Figure 3.10B), and Wood Thrush ($P < 0.001$, $\beta = -0.041$, SE = 0.010; Figure 3.10E). As the breeding season progressed, detection probability increased for Kentucky Warbler but decreased for Common Yellowthroat and Wood Thrush. In addition, increasing percent of cloud cover had a significant negative influence on detection probability for Prairie Warbler ($P < 0.001$, $\beta = -0.005$, SE = 0.001; Figure 3.10C), Swainson's Warbler ($P = 0.05$, $\beta = -0.007$, SE = 0.003; Figure 3.10D), and Wood Thrush ($P < 0.01$, $\beta = 0.006$, SE = 0.002).

Projected yearly occupancy varied by species (Figure 3.11), with the two early successional species decreasing and the three woodland species increasing in projected occupancy in response to time. Common Yellowthroat occupancy remained greater than 0.85 for the entire 11 years across all sites, but occupancy probability decreased significantly ($P < 0.001$) in the Pine Control plots over time when compared to Switchgrass Intercropped Plots (Figure 3.12A). Kentucky Warbler occupancy increased from 2011 to 2021, regardless of treatment ($P =$

0.135; Figure 3.12B), while Prairie Warbler projected occupancy peaked in years three through six of the study (Figure 3.12C), decreasing to a 0.12 occupancy probability in 2021 irrespective of treatment ($P = 0.58$). Swainson's Warbler projected occupancy remained less than 0.3 during the entire study but increased in projected values from 2011 to 2021. Treatment significantly influenced Swainson's Warbler projected occupancy, with Pine Control plots exhibiting greater projected probabilities than Switchgrass Intercropped plots ($P < 0.001$; Figure 3.12D). Similarly, Wood Thrush projected occupancy increased from 2011 to 2021, with increases independent of treatment ($P = 0.148$; Figure 3.12E).

2011–2021 Avian Community Structure

NMDS analysis for 2019–2021 showed less overall variation in avian community structure between years, with more centralized and small ellipses (Figure 3.13a). There was significant response to survey year ($R^2 = 0.77$ and $P < 0.05$), but not to treatment ($R^2 = 0.16$ and $P = 0.63$). NMDS by treatment analysis (Figure 3.13b) showed considerable overlap in the avian community among treatments across the 3-year period. NMDS illustrated clustering by species, specifically woodland species, with similar scoring along both axes.

NMDS analysis for 2011-2021 showed a clear separation between stages (Figure 3.14). There was a significant response to treatment by stage ($R^2 = 0.41$ and $P = 0.001$). Treatments within stages had considerable overlap, with no overlap between stages. Avian species were generally clustered by known habitat association, with separations between woodland, early successional/scrub, and grassland species.

Discussion

Convergence of bird communities occurred between treatments as forest succession progressed in this system, with communities becoming increasingly similar as the forest stands aged. While bird diversity often decreases in response to increasingly dense understories (Dickson et al. 1993), species diversity remained high throughout the later years of this study. This could possibly be explained by the differences in the vegetation community, as a result of frequent disturbances (e.g., human disturbance during surveys and recreational use, mechanical clearing of trails, repeated seeding of switchgrass) that created small forest openings mimicking natural disturbances. Over time, increases in shrub and vine cover likely created higher levels of nesting substrates for many early successional and scrub species (Lohr et al. 2002, Capel et al. 1994), leading to the observed high detections of these species even as the forest canopy and shading of the understory increased. These high diversity levels lead to renewed increases in CV within all treatments.

Intercropping switchgrass between pine rows and implementing hardwood control through chemical application only marginally influenced bird community composition within my study years, with more demonstratable effects during the 11-year composite study (Murray and Best 2003 but see Loman et al. 2014). Comparisons of one metric, specifically either diversity or CV, did not provide much insight into the subtle changes observed. However, when comparing the predicted densities of individual species, there were strong responses elicited by both management techniques. For modeled species across my study period, there were eight instances of five species with higher estimated densities within Hardwood Control treatments compared to Pine Control treatments, and ten instances of six species with higher densities in Switchgrass Intercropped plots when compared to Pine Control plots. Conversely, there was one species, Hooded Warbler, with a negative density response to switchgrass intercropping, and one species, Yellow-billed Cuckoo, with a negative density response to hardwood control.

There was considerable species-specific variation in model fitting and predictive power for density estimates. With species in this system often having effective detection radii of over 50m (Iglay et al. 2012), models using point count survey distances of only up to 50m could create model fitting issues because of the lack in the complete detection distribution. This could also be due to small sample sizes, inaccurate distance detections, or affinity for particular vegetation characteristics not evenly spaced throughout each treatment. In fact, Buckland et. al (2001) recommended only creating distance-based abundance estimates for species with at least 75–100 detections per season, a standard sample size that is reached often only by the most common species in many southeastern pine forests (see Table 3.5; personal data, Legrand et al. 2007, Marshall 2016, Parrish et al. 2017, Fuller-Morris 2018). In a similar fashion, while there were significant differences between the response diversity and CV between the various species guilds used, it is important to recognize that sample size within guilds greatly affects estimates. With a non-equal number of species within guilds, and consequently a non-equal effect of PIF ranks, comparisons across guilds must be interpreted carefully if completed.

Use of imazapyr and metsulfuron-methyl as a means of controlling hardwood, *Rubus* spp, and non-native vegetation density appears to have limited impacts beyond the application season (Bohn et al. 2011, Albaugh et al. 2012). Previous studies confirmed that herbicide application limited hardwood and shrub cover within the applied areas in the years immediately following use (Quicke et al. 1996, Cheynet 1999, Albaugh et al. 2003); this control can then lead to an increase in forbs and grasses (Fuller-Morris 2018) and other desirable herbaceous understory vegetation for wildlife (Singleton et al. 2013). However, beginning four years after application, the amount of shrub cover rebounded in areas previously treated to levels higher than in Pine Control plots. Similarly, there was only a difference in amount of hardwood cover in one of three years of this study, with less hardwood in the treated plots when compared to the Pine Control plots in 2020. These changes likely led to fewer differences in amount of forb and grass cover across time between Pine Control and treatment plots. Increasing percent cover of shrubs possibly explains the decreases in grasses when compared to previous years (Fuller-Morris 2018) and observed increases in understory densities. In all likelihoods, the commonality that results from managing for one specific dominant tree species regardless of treatment, proximity of treatments within each stand or site, and planting of trees in comparable ages resulting in trees of equivalent size, height, and canopy structure that create similar ambient light patterns may control the trajectory of the understory vegetation (Smith et al. 1994, del Moral 2007, Barbier et al. 2008, Ádám et al. 2013, Kermavnar et al. 2019). However, establishment of temporal and spatial scales within the sampling design employed determines the ability to elucidate these patterns (Lepš and Rejmánek 1991). Overall, as changes within the communities studied are a continuum, vegetation changes within all treatments likely created the observed overlaps in avian community structure across time.

The increasing percent of shrub coverage post hardwood control implementation rose to levels beyond the Pine Control plots within several years of chemical control, with impacts of such treatment having little effect on the avian community. While it is well established that site preparation prior to stand initiation has documented impacts on the subsequent avian community, less is known about effects of experimental active management after activity has ceased (Lane et al. 2011, Greene et al. 2016). Limited research has indicated that site preparation, or harvest disturbance, effects decrease across time (Hunt et al. 2003, Iglay et al. 2012). Treatment effects within the vegetation community similarly decrease post site preparation, with vegetation characteristics quickly overlapping between treatments (Iglay et al. 2012). Clear interactions exist between understory vegetation structure, pine structure, and the corresponding bird community, but less is known about long-term implications of management within aging pine stands (Kilgo et al. 2000, Owens et al. 2014).

Perhaps more important than these two applied management techniques, switchgrass intercropping and hardwood control, on avian community composition in this system is time since stand establishment and time since management. Comparisons between years, both with and without a treatment interaction, indicated that time since establishment likely plays a role in observed community structure (Greene et al. 2016). Analyses comparing responses of all species, migration guilds, and habitat guilds produced significant CV responses to increasing time since establishment across the 11-year study. Year 11 of this study produced many woodland species, possibly indicating a similarity to later forest stages in response to vegetation changes and a closing canopy. While canopy cover estimates have been shown to have high predictive power in determining occupancy of particular species (James 1971, Hinsley et al. 2009), it was less useful in occupancy models in this study as measurements were only taken during the final four years of the project and had a non-significant influence on species occupancy when included.

With previous research indicating that switchgrass planted within pine stands leads to increases in labile carbon, respiration, and microbial biomass and activity levels (Blazier et al. 2012), further research is needed to determine the utilities and trade-off of this method as stands age and the canopy closes. If loblolly pine stands are being managed primarily for timber production, intercropping switchgrass may only be a useful biofuel crop for a few years following stand initiation (Tian et al. 2017). Similarly, if switchgrass management ceases, switchgrass plots may only support high abundances of some species until such time that the vegetation structure converges with standard management, as demonstrated by the decreasing number of species with abundances differences between Pine Control and Switchgrass Intercropped plots in this study. But opportunities exist to investigate the relationship between long term switchgrass intercropping, pine management, and the avian community, as this research has clearly documented the ability of several avian species to maintain higher abundances in switchgrass intercropped plots compared to Pine Control plots for 6 years post management.

Conclusion and Management Implication

Ecological effects of both intercropping switchgrass and controlling hardwood densities by herbicide application may be negligible across time as understory structure and composition rebound after management. Time since stand establishment had a more clearly defined relationship with avian community composition and abundances of many bird species. But understory vegetation composition and structure cannot be ignored as an essential component of forest ecosystem management considerations. If bird conservation and management is a primary focus in a working forest, managers should consider both short- and long-term effects of implemented decisions. When focusing on early successional species, many of which are of high conservation concern, it becomes important to maintain vegetation conditions that promote high breeding densities and reproductive success, including creating areas of less canopy cover within aging stands that would promote understory vegetation complexity and grassland conditions.

Hardwood control and switchgrass intercropping appear to have had limited long-term implications on the bird community, though some species-specific and yearly consequences exist. Within the large, dynamic landscape, species that may exhibit negative or even neutral responses to the particular management techniques used in this study generally have the ability to move to a forest stand more proximate to more suitable conditions. As such, when implementing non-traditional management methods such as intercropping, it would be advised to scatter intercropping across the landscape and not intercrop adjacent stands. To reach desirable vegetation conditions for wildlife, prescribed fire should also be considered in conjunction with herbicide use. Further research should not only focus on movements of individuals within, between, and out of managed stands, but also on using these management methods at various times during a stand rotation, promoting understory structure as a means of providing appropriate breeding habitat for early successional species even as stands age and the canopy closes, and evaluating annual avian reproductive success and survivorship in response to management.

Tables and Figures

Table 3.1 Daubenmire cover classes used to report average percent cover of plant guilds and understory density in managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi.

Daubenmire Score[a]	Percent Cover	Midpoint
1	0–5%	2.5%
2	5–25%	15%
3	25–50%	37.5%
4	50–75%	62.5%
5	75–95%	85%
6	95–100%	97.5%

[a] Score class values from Daubenmire (1959).

Table 3.2 List of all breeding avian species detected during point count surveys May-June 2019–2021 within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi, with the associated American Ornithological Society/Institute for Bird Populations alpha code (Pyle and DeSante 2003), modified Partners in Flight ranks, migratory status, and known breeding habitat association.

Species		AOU Code	PIF Score[a]	Migratory Status[b]	Habitat[c]
Acadian Flycatcher	*Empidonax virescens*	ACFL	2	N	W
American Crow	*Corvus brachyrhynchos*	AMCR	1	R	W
American Goldfinch	*Spinus tristis*	AMGO	1	S	ES/S
Barred Owl	*Strix varia*	BARO	1	R	W
Black-and-white Warbler	*Mniotilta varia*	BAWW	1	N	W
Blue Grosbeak	*Passerina caerulea*	BLGR	1	N	W
Blue Jay	*Cyanocitta cristata*	BLJA	1	N	ES/S
Blue-gray Gnatcatcher	*Polioptila caerulea*	BGGN	1	S	U
Broad-winged Hawk	*Buteo platypterus*	BWHA	1	S	W
Brown-headed Cowbird	*Molothrus afer*	BHCO	1	S	G
Brown-headed Nuthatch	*Sitta pusilla*	BHNU	3	R	W
Brown Thrasher	*Toxostoma rufum*	BRTH	2	N	ES/S
Carolina Chickadee	*Poecile carolinensis*	CACH	1	R	W
Carolina Wren	*Thryothorus ludovicianus*	CARW	1	R	ES/S
Chipping Sparrow	*Spizella passerina*	CHSP	1	R	U
Common Grackle	*Quiscalus quiscula*	COGR	1	S	U
Common Nighthawk	*Chordeiles minor*	CONI	3	N	G
Common Yellowthroat	*Geothlypis trichas*	COYE	1	N	ES/S
Cooper’s Hawk	*Accipiter cooperii*	COHA	1	N	W
Downy Woodpecker	*Dryobates pubescens*	DOWO	1	R	W
Eastern Bluebird	*Sialia sialis*	EABL	1	R	G
Eastern Kingbird*	*Tyrannus tyrannsu*	EAKI	3	N	G
Eastern Towhee*	*Pipilo erythrophthalmus*	EATO	3	S	ES/S
Eastern Wood-Pewee*	*Contopus virens*	EAWP	3	N	W
Field Sparrow*	*Spizella pusilla*	FISP	3	S	ES/S
Fish Crow	*Corvus ossifragus*	FICR	1	R	WE
Gray Catbird	*Dumetella carolinensis*	GRCA	1	N	ES/S
Great Crested Flycatcher	*Myiarchus crinitus*	GCFL	1	N	W
Hairy Woodpecker	*Dryobates villosus*	HAWO	1	R	W
Hooded Warbler	*Setophaga citrina*	HOWA	1	N	W
Indigo Bunting*	*Passerina cyanea*	INBU	4	N	ES/S
Kentucky Warbler*	*Geothlypis formosa*	KEWA	4	N	W

Table 3.2 (continued)

Species		AOU Code	PIF Score[a]	Migratory Status[b]	Habitat[c]
Killdeer	*Charadrius vociferus*	KILL	1	S	G
Mourning Dove	*Zenaida macroura*	MODO	1	S	U
Northern Bobwhite*	*Colinus virginianus*	NOBO	4	R	ES/S
Northern Cardinal	*Cardinalis cardinalis*	NOCA	1	R	ES/S
Northern Flicker	*Colaptes auratus*	NOFL	2	S	W
Northern Mockingbird	*Mimus polyglots*	NOMO	1	R	U
Northern Parula	*Setophaga americana*	NOPA	1	N	W
Ovenbird	*Seiurus aurocapilla*	OVEN	2	N	W
Pileated Woodpecker	*Dryocopus pileatus*	PIWO	1	R	W
Pine Warbler	*Setophaga pinus*	PIWA	1	R	W
Prairie Warbler*	*Setophaga discolor*	PRAW	3	N	ES/S
Purple Martin	*Progne subis*	PUMA	3	N	U
Red-bellied Woodpecker	*Melanerpes carolinus*	RBWO	1	R	W
Red-eyed Vireo	*Vireo olivaceus*	REVI	1	N	W
Red-headed Woodpecker*	*Melanerpes erythrocephalus*	RHWO	3	R	W
Red-shouldered Hawk	*Buteo lineatus*	RSHA	1	S	W
Ruby-throated Hummingbird	*Archilochus colubris*	RTHU	1	N	W
Scarlet Tanager	*Piranga olivacea*	SCTA	2	N	W
Summer Tanager	*Piranga rubra*	SUTA	1	N	W
Swainson's Warbler*	*Limnothlypis swainsonii*	SWWA	1	N	W
Tufted Titmouse	*Baeolophus bicolor*	TUTI	1	R	W
Turkey Vulture	*Cathartes aura*	TUVU	1	R	W
White-eyed Vireo	*Vireo griseus*	WEVI	1	N	ES/S
Wood Thrush*	*Hylocichla mustelina*	WOTH	4	N	W

*Denotes priority bird classification by the Eastern Gulf Coastal Plain Joint Venture Landbird Conservation Plan (Greene et al. 2021).

[a] PIF Rank calculated from the Partners in Flight Avian Conservation Assessment Database, using methodology of Nuttle et al. (2003).

[b] Migratory status abbreviations: N- neotropical migrant, S- short distance migrant, R- resident; retrieved from the Breeding Bird Survey database (Sauer et al. 2017).

[c] Associated habitat type abbreviations: ES/S - early successional/scrub, G - grassland, U - urban, WE - wetland, W - woodland; retrieved from the Breeding Bird Survey database (Sauer et al. 2017).

Table 3.3 The set of dynamic (colonization-extinction) occupancy models fit to avian point count data within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi, in May and June 2011–2021. Additive covariate relationships are indicated by + (i.e., shrub + forbs). Model selection was based on Akaike's Information Criterion (AIC) and corrected for overdispersion and small sample size, yielding quasi-AICc (QAICc) values. Models were ranked by QAICc, and the number of parameters (K), ΔQAICc, model weight (W_i), cumulative model weight (Cum. W_i), and relative model likelihood (Q Lk.hd.) were reported.

Species	**Model**	**K**	**QAICc**	**ΔQAICc**	**W_i**	**Cum.W_i**	**Q Lk.hd.**
Common Yellowthroat	~ψ(.) ~γ(.) ~ε(.) ~p(obs + jdate)	18	979.99	0.00	0.94	0.94	-458.84
	~ψ(.) ~γ(.) ~ε(vine) ~p(obs + jdate)	19	985.45	5.45	0.06	1.00	-458.52
	~ψ(.) ~γ(vine) ~ε(vine) ~p(obs + jdate)	20	991.45	11.46	0.00	1.00	-458.22
	~ψ(.) ~γ(.) ~ε(.) ~p(.)	5	1096.19	116.2	0.00	1.00	-542.33
Prairie Warbler	~ψ(.) ~γ(.) ~ε(shrub) ~p(cloud)	7	760.86	0.00	0.96	0.96	-371.91
	~ψ(.) ~γ(.) ~ε(.) ~p(.)	5	768.26	7.41	0.02	0.98	-378.36
	~ψ(.) ~γ(.) ~ε(shrub) ~p(.)	6	770.70	9.85	0.01	0.99	-378.25
	~ψ(.) ~γ(forbs) ~ε(shrub) ~p(.)	7	772.96	12.10	0.00	0.99	-377.97
	~ψ(.) ~γ(.) ~ε(shrub + forbs) ~p(.)	7	773.00	12.14	0.00	0.99	-377.99
	~ψ(forbs) ~γ(.) ~ε(shrub) ~p(.)	7	773.00	12.15	0.00	0.99	-377.99
	~ψ(.) ~γ(allgrass) ~ε(shrub) ~p(.)	7	773.03	12.17	0.00	1.00	-378.00
	~ψ(allgrass) ~γ(allgrass) ~ε(shrub) ~p(.)	7	773.27	12.42	0.00	1.00	-378.12
	~ψ(.) ~γ(.) ~ε(shrub + allgrass) ~p(.)	7	773.39	12.53	0.00	1.00	-378.18
Swainson's Warbler	~ψ(.) ~γ(forbs) ~ε(forbs) ~p(cloud)	7	222.67	0.00	0.52	0.52	-102.82
	~ψ(.) ~γ(.) ~ε(forbs) ~p(cloud)	8	224.38	1.70	0.22	0.74	-102.19
	~ψ(.) ~γ(shrub) ~ε(forbs) ~p(cloud)	8	224.63	1.96	0.19	0.93	-102.31
	~ψ(.) ~γ(forbs + shrub) ~ε(forbs) ~p(cloud)	9	226.79	4.12	0.07	1.00	-101.82
	~ψ(.) ~γ(.) ~ε(.) ~p(.)	5	236.54	13.86	0.00	1.00	-112.50

Table 3.3 (continued)

Species	**Model**	**K**	**QAICc**	**ΔQAICc**	**W_i**	**Cum.W_i**	**Q Lk.hd.**
Wood Thrush	~ψ(.) ~γ(.) ~ε(.) ~p(cloud + jdate)	7	535.37	0.00	0.29	0.29	-259.17
	~ψ(.) ~γ(.) ~ε(graminoid) ~p(cloud + jdate)	8	537.25	1.88	0.11	0.40	-258.62
	~ψ(.) ~γ(vine) ε(.) ~p(cloud + jdate)	8	537.35	1.98	0.11	0.51	-258.68
	~ψ(.) ~γ(.) ~ε(allgrass) ~p(cloud + jdate)	8	537.69	2.32	0.09	0.60	-258.84
	~ψ(.) ~γ(.) ε(switchgrass) ~p(cloud + jdate)	8	537.78	2.41	0.09	0.69	-258.89
	~ψ(.) ~γ(.) ε(forbs) ~p(cloud + jdate)	8	537.83	2.46	0.08	0.77	-258.92
	~ψ(.) ~γ(allgrass) ~ε(.) ~p(cloud + jdate)	8	538.15	2.78	0.07	0.84	-259.07
	~ψ(.) ~γ(vine) ~ε(graminoid) ~p(cloud + jdate)	9	539.30	3.94	0.04	0.88	-258.08
	~ψ(.) ~γ(.) ~ε(graminoid+shrub) ~p(cloud + jdate)	9	539.47	4.10	0.04	0.92	-258.16
	~ψ(.) ~γ(.) ~ε(graminoid+vine) ~p(cloud + jdate)	9	539.66	4.29	0.03	0.95	-258.26
	~ψ(.) ~γ(allgrass) ~ε(graminoid) ~p(cloud + jdate)	9	540.18	4.81	0.03	0.98	-258.52
	~ψ(.) ~γ(forbs) ~ε(vine) ~p(cloud + jdate)	9	540.69	5.32	0.02	1.00	-258.77
	~ψ(.) ~γ(.) ~ε(.) ~p(.)	5	556.56	21.19	0.00	1.00	-272.51

Ψ – psi: formula for initial probability of occupancy
γ – gamma: formula for colonization probability
ε – epsilon: formula for extinction probability
p – formula for detection probability

Table 3.4 The set of dynamic (colonization-extinction) occupancy models fit to avian point count data within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi, in May and June 2011–2021. Additive covariate relationships are indicated by + (i.e., shrub + forbs). Model selection was based on Akaike's Information Criterion (AIC). Models were ranked by AICc, and the number of parameters (K), ΔAICc, model weight (W_i), cumulative model weight (Cum. W_i), and model log likelihood (LogLik) were reported.

Species	**Model**	**K**	**AICc**	**ΔAICc**	**W_i**	**Cum.W_i**	**LogLik**
Kentucky Warbler	~ψ(.) ~γ(.) ~ε(.) ~p(jdate+obs)	17	904.06	0	0.41	0.41	-435.029
	~ψ(.) ~γ(.) ~ε(vine) ~p(jdate+obs)	18	904.84	0.78	0.28	0.68	-434.43
	~ψ(.) ~γ(.) ~ε(graminoid) ~p(jdate+obs)	18	905.94	1.88	0.16	0.84	-434.969
	~ψ(.) ~γ(graminoid) ~ε(.) ~p(jdate+obs)	18	905.96	1.9	0.16	1.00	-434.982
	~ψ(.) ~γ(.) ~ε(.) ~p(.)	4	1052	147.94	0.00	1.00	-522.109

Ψ – psi: formula for initial probability of occupancy
γ – gamma: formula for colonization probability
ε – epsilon: formula for extinction probability
p – formula for detection probability

Table 3.5 Number of detections for each bird species detected, by year and treatment, in May and June 2019–2021, within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. Flyovers, flythroughs, and detections of females are removed.

Species	2019				2020				2021			
	C	HC	SI	Total	C	HC	SI	Total	C	HC	SI	Total
Acadian Flycatcher	0	0	0	0	0	2	0	2	0	0	0	0
American Crow	4	1	4	9	4	5	4	13	0	0	0	0
American Goldfinch	1	0	0	1	0	0	0	0	0	0	0	0
Barred Owl	0	0	0	0	0	2	0	2	0	0	0	0
Black-and-white Warbler	17	2	6	25	22	17	21	60*	24	13	16	53*
Blue Grosbeak	1	3	7	11	4	6	3	13	0	3	1	4
Blue Jay	10	12	16	38	47	64	53	164*	29	34	28	91*
Blue-gray Gnatcatcher	7	4	2	13	2	0	2	4	1	4	2	7
Broad-winged Hawk	0	0	0	0	0	0	1	1	0	0	2	2
Brown Thrasher	0	1	2	3	10	3	10	23	4	4	7	15
Brown-headed Cowbird	9	6	31	46*	8	1	6	15	8	10	16	34
Brown-headed Nuthatch	0	0	0	0	0	0	0	0	0	1	0	1
Carolina Chickadee	13	8	8	29	13	7	26	46*	2	11	9	22
Carolina Wren	28	67	43	138*	41	70	59	170*	56	83	69	208*
Chipping Sparrow	1	2	2	5	4	1	3	8	0	0	0	0
Common Grackle	0	0	0	0	0	0	0	0	0	1	0	1
Common Nighthawk	0	0	0	0	0	0	0	0	1	0	0	1
Common Yellowthroat	24	61	79	164*	7	69	49	125*	7	17	19	43*
Cooper's Hawk	1	0	0	1	0	0	0	0	0	0	0	0
Downy Woodpecker	0	0	2	2	1	4	2	7	4	2	0	6
Eastern Bluebird	0	0	0	0	1	0	0	1	0	0	0	0
Eastern Kingbird	0	0	0	0	0	0	0	0	0	1	0	1
Eastern Towhee	59	110	111	280*	61	104	91	256*	32	50	60	142*
Eastern Wood-Pewee	0	1	0	1	0	0	0	0	0	0	0	0
Field Sparrow	0	0	1	1	0	0	1	1	0	1	1	2
Fish Crow	0	0	1	1	2	4	0	6	0	0	0	0
Gray Catbird	4	13	13	30	4	8	11	23	0	2	2	4
Great Crested Flycatcher	11	6	6	23	13	10	9	32	23	15	7	45*
Hairy Woodpecker	0	0	0	0	0	1	0	1	0	0	0	0
Hooded Warbler	54	48	6	108*	77	83	57	217*	73	62	39	174*
Indigo Bunting	16	24	40	80*	5	34	33	72*	10	18	31	59*
Kentucky Warbler	15	6	2	23	42	44	23	109*	33	43	36	112*
Killdeer	0	0	0	0	1	0	0	1	0	0	0	0
Mourning Dove	4	4	2	10	24	23	22	69*	26	10	13	49*
Northern Bobwhite	3	1	2	6	22	14	18	54*	11	0	8	19

Table 3.5 (continued)

Species	2019				2020				2021			
	C	HC	SI	Total	C	HC	SI	Total	C	HC	SI	Total
Northern Cardinal	51	58	58	167*	107	86	83	276*	64	44	39	147*
Northern Flicker	0	1	0	1	4	2	2	8	1	2	0	3
Northern Mockingbird	0	1	4	5	0	0	1	1	0	0	0	0
Northern Parula	0	1	0	1	0	1	0	1	0	1	0	1
Orchard Oriole	4	2	8	14	1	0	3	4	2	1	6	9
Ovenbird	1	0	0	1	0	0	0	0	0	0	0	0
Pileated Woodpecker	0	0	0	0	3	2	5	10	4	6	1	11
Pine Warbler	13	30	26	69*	14	26	26	66*	2	10	15	27
Prairie Warbler	9	8	21	38	2	5	13	20	4	4	4	12
Red-bellied Woodpecker	1	6	3	10	31	17	19	67*	13	3	7	23
Red-eyed Vireo	4	5	15	24	3	2	5	10	6	3	5	14
Red-headed Woodpecker	1	0	0	1	5	1	6	12	2	0	1	3
Red-shouldered Hawk	0	0	0	0	0	1	0	1	0	0	0	0
Ruby-throated Hummingbird	1	1	3	5	1	3	2	6	1	1	5	7
Scarlet Tanager	0	0	0	0	1	0	1	2	0	0	0	0
Summer Tanager	2	1	4	7	6	4	9	19	2	6	12	20
Swainson's Warbler	13	0	1	14	13	2	7	22	8	1	0	9
Tufted Titmouse	0	1	1	2	2	3	5	10	8	5	1	14
White-eyed Vireo	183	128	190	501*	109	104	126	339*	42	44	99	185*
Wild Turkey	0	0	0	0	0	1	1	2	0	0	0	0
Wood Thrush	13	18	13	44*	36	58	65	159*	51	35	53	139*
Worm-eating Warbler	0	0	0	0	8	1	0	9	25	5	0	30
Yellow-billed Cuckoo	8	4	6	18	26	27	30	83*	25	6	12	43*
Yellow-breasted Chat	100	148	162	410*	98	122	168	388*	39	46	92	177*
Yellow-throated Vireo	1	0	5	6	0	2	7	9	0	2	0	2
Total	687	793	906	2386	885	1046	1088	3019	644	610	718	1972

Abbreviations: C – Pine Control, HC - Hardwood Control, SI - Switchgrass Intercropped
*Notes use in density estimate analysis

Table 3.6 Avian species richness, Shannon's diversity indices, and Conservation Value (CV) among hardwood control treated loblolly pine (*Pinus taeda*) stands, switchgrass (*Panicum virgatum*) intercropped loblolly pine stands, and stands with no additional management in east-central Mississippi.

		Treatment			
	Year	Pine Control	Hardwood Control	Switchgrass Intercropped	Total
Richness	2019	34	36	38	45
	2020	42	44	43	52
	2021	35	40	34	44
Diversity	2019	2.65	2.61	2.64	2.70
	2020	3.01	2.96	3.04	3.05
	2021	3.03	3.00	2.92	3.06
CV	2019	1.44	1.50	1.51	1.49
	2020	1.59	1.70	1.66	1.65
	2021	1.70	1.68	1.78	1.72

Table 3.7 ANOVA results from comparisons of Shannon's diversity index and Conservation Value among hardwood control treated loblolly pine (*Pinus taeda*) stands, switchgrass (*Panicum virgatum*) intercropped loblolly pine stands, and stands with no additional management in east-central Mississippi, for birds grouped by migration status and known breeding habitat association. Bolded P-values indicate statistical significance (α = 0.05).

Group		Metric		F-value	P-value
All species		Diversity	Year	10.58	**0.022**
			Treatment	0.08	0.914
		CV	Year	39.65	**0.001**
			Treatment	2.11	0.234
Migration	Resident	Diversity	Year	0.09	0.779
			Treatment	0.25	0.791
		CV	Year	1.04	0.355
			Treatment	1.33	0.344
	Short Distance	Diversity	Year	33.38	**0.002**
			Treatment	2.70	0.160
		CV	Year	23.64	**0.005**
			Treatment	1.62	0.288
	Neotropical	Diversity	Year	31.76	**0.002**
			Treatment	0.35	0.700
		CV	Year	142.53	< **0.001**
			Treatment	0.22	0.81
Habitat	Early successional/Scrub	Diversity	Year	3.84	0.108
			Treatment	2.47	0.179
		CV	Year	1.48	0.278
			Treatment	11.04	**0.015**
	Woodland	Diversity	Year	0.00	0.966
			Treatment	1.51	0.306
		CV	Year	26.95	**0.003**
			Treatment	0.53	0.621

Table 3.8 Predicted male densities (birds per hectare) ± standard errors per treatment plot for breeding bird species with at least 40 detections within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi between May and June 2019. Bolded estimates indicate significant differences ($\alpha = 0.05$) in abundance estimates between treatment and control plots.

	Treatment					
Species	Pine Control	95% CI	Hardwood Control	95% CI	Switchgrass Intercropped	95% CI
Brown-headed Cowbird	24.44 ± 38.51	1.11 – 536.52	16.54 ± 25.96	0.76 – 358.23	**84.46 ± 128.89**	**4.24 – 1681.09**
Carolina Wren	7.06 ± 5.96	1.35 – 36.97	**17.17 ± 14.15**	**3.36 – 86.69**	10.63 ± 9.04	2.01 – 56.31
Common Yellowthroat	1.72 ± 0.54	0.93 – 3.18	**4.18 ± 1.15**	**2.44 – 7.16**	**5.30 ± 1.36**	**3.21 – 8.76**
Eastern Towhee	10.14 ± 8.96	1.80 – 57.32	**18.07 ± 16.26**	**3.10 – 105.36**	**18.68 ± 16.63**	**3.26 – 106.97**
Hooded Warbler	9.35 ± 6.03	2.64 – 33.13	8.24 ± 5.31	2.33 – 29.13	**1.27 ± 0.83**	**0.36 – 4.56**
Indigo Bunting	1.06 ± 0.42	0.49 – 2.31	1.63 ± 0.56	0.83 – 3.19	**2.88 ± 0.89**	**1.58 – 5.26**
Pine Warbler	4.34 ± 2.24	0.64 – 29.48	**10.60 ± 9.82**	**1.72 – 65.14**	8.99 ± 8.46	1.42 – 56.82
Yellow-breasted Chat	3.94 ± 0.74	2.72 – 5.69	5.63 ± 1.00	3.97 – 7.98	**6.26 ± 1.06**	**4.49 – 8.72**

Table 3.9 Predicted male densities (birds per hectare) ± standard errors per treatment plot for breeding bird species with at least 40 detections within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi between May and June 2020. Bolded estimates indicate significant differences (α = 0.05) in abundance estimates between treatment and control plots.

	Treatment					
Species	Pine Control	95% CI	Hardwood Control	95% CI	Switchgrass Intercropped	95% CI
Carolina Wren	1.54 ± 1.25	0.32 – 7.56	**2.68 ± 2.21**	0.54 – 13.46	2.24 ± 1.85	0.44 – 11.31
Common Yellowthroat	0.50 ± 0.82	0.02 – 12.83	**4.11 ± 7.28**	**0.13 – 132.29**	**3.07 ± 5.38**	**0.10 – 95.35**
Eastern Towhee	1.31 ± 0.23	0.92 – 1.86	**2.04 ± 0.31**	**1.51 – 2.74**	1.80 ± 0.29	1.31 – 2.47
Indigo Bunting	0.06 ± 0.04	0.01 – 0.24	**0.39 ± 0.17**	**0.17 – 0.92**	**0.55 ± 0.20**	**0.27 – 1.12**
Yellow-breasted Chat	1.57 ± 0.25	1.15 – 2.15	1.99 ± 0.28	1.50 – 2.63	**2.74 ± 0.34**	**2.14 – 3.50**

Table 3.10 Predicted male densities (birds per hectare) ± standard errors per treatment plot for breeding bird species with at least 40 detections within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi between May and June 2021. Bolded estimates indicate significant differences (α = 0.05) in abundance estimates between treatment and control plots.

	Treatment					
Species	Pine Control	95% CI	Hardwood Control	95% CI	Switchgrass Intercropped	95% CI
Great Crested Flycatcher	21.55 ± 24.25	2.37 – 195.53	15.43 ± 17.99	1.57 – 151.60	**6.74 ± 8.10**	**0.64 – 70.96**
Hooded Warbler	22.35 ± 17.59	4.78 – 104.52	19.30 ± 15.51	4.00 – 93.20	**12.17 ± 9.83**	**2.50 – 59.27**
White-eyed Vireo	2.49 ± 0.48	1.70 – 3.63	2.57 ± 0.49	1.76 – 3.73	**5.91 ± 0.81**	**4.52 – 7.74**
Yellow-billed Cuckoo	2.18 ± 2.56	0.22 – 21.85	**0.55 ± 0.65**	**0.05 – 5.52**	1.12 ± 1.25	0.12 – 10.05
Yellow-breasted Chat	0.96 ± 0.21	0.62 – 1.48	1.10 ± 0.23	0.73 – 1.66	**2.21 ± 0.35**	**1.62 – 3.02**

Table 3.11 Predicted male densities (males per hectare) ± standard errors for breeding bird species, with at least 40 detections and no treatment level effects, within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi between May and June 2021.

Year	Species	Estimate (males/hectare)	95% CI
2019	Northern Cardinal	22.81 ± 43.18	0.56 – 932.52
	White-eyed Vireo	7.20 ± 1.33	5.01 – 10.34
	Wood Thrush	22.55 ± 17.21	5.05 – 100.61
2020	Black-and-white Warbler	20.88 ± 23.54	2.29 – 190.32
	Blue Jay	21.27 ± 21.70	2.88 – 157.08
	Carolina Chickadee	1.26 ± 2.44	0.03 – 55.43
	Hooded Warbler	2.32 ± 0.26	1.87 – 2.89
	Kentucky Warbler	0.62 ± 0.10	0.45 – 0.84
	Mourning Dove	1.55 ± 2.48	0.07 – 35.54
	Northern Bobwhite	20.24 ± 32.13	0.90 – 454.41
	Northern Cardinal	1.63 ± 0.18	1.32 – 2.02
	Red-bellied Woodpecker	1.22 ± 1.24	0.17 – 8.94
	White-eyed Vireo	3.91 ± 0.34	3.30 – 4.64
	Yellow-billed Cuckoo	1.47 ± 1.11	0.33 – 6.44
2021	Black-and-white Warbler	1.74 ± 2.57	0.10 – 31.44
	Blue Jay	20.73 ± 27.26	1.57 – 272.88
	Carolina Wren	1.95 ± 0.22	1.56 – 2.44
	Common Yellowthroat	0.24 ± 0.07	0.14 – 0.43
	Eastern Towhee	3.46 ± 4.89	0.22 – 55.30
	Indigo Bunting	0.34 ± 0.08	0.22 – 0.53
	Kentucky Warbler	1.19 ± 0.18	0.89 – 1.61
	Mourning Dove	0.45 ± 0.18	0.20 – 0.99
	Northern Cardinal	2.26 ± 0.29	1.76 – 2.92
	Wood Thrush	0.87 ± 0.13	0.65 – 1.16

Table 3.12 Models contributing to at least 0.90 of the cumulative model weight for predicting occupancy for five avian species within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi from 2011–2021 using dynamic (colonization-extinction) occupancy models. For each model, covariates are listed with parameter estimates (original logit scale) and 95% confidence (UCI, UCI) intervals in parentheses. Site parameters include percent cover of vine, graminoid, shrub, and forbs and detection parameters include percent cloud and Julian date (jdate). Observer parameter coefficients not included. Bolded parameters indicate statistical significance (α = 0.05).

		Covariate (estimate [UCI, LCI])			
Species	**Model**	**ψ (Initial Occupancy)**	**γ (Colonization)**	**ε (Extinction)**	**p (Detection)**
Common Yellowthroat	1				**jdate (-0.147 [-0.254, -0.039])**
Kentucky Warbler	1				**jdate (0.058 [0.030, 0.085])**
	2			vine (0.081 [-0.059, 0.221])	**jdate (0.058 [0.030, 0.085])**
	3			graminoid (-0.019 [-0.112, 0.074])	**jdate (0.058 [0.030, 0.085])**
	4		graminoid (0.031 [-0.173, 0.234])		**jdate (0.058 [0.030, 0.085])**
Prairie Warbler	1			shrub (-0.075 [-0.247, 0.097])	**cloud (-0.005 [-0.008, -0.003])**
Swainson's Warbler	1		forbs (0.034 [-0.004, 0.072])	**forbs (-0.126 [-0.219, -0.033])**	**cloud (-0.007 [-0.013, 0.000])**
	2			**forbs (-0.129 [-0.221, -0.036])**	**cloud (-0.007 [-0.013, 0.000])**
	3		shrub (0.179 [-0.034, 0.392])	**forbs (-0.128 [-0.221, -0.035])**	**cloud (-0.007 [-0.013, 0.000])**
Wood Thrush	1				**cloud (0.006 [0.002, 0.010])** **jdate (-0.041 [-0.059, -0.021])**
	2			graminoid (-0.055 [-0.125, 0.015])	**cloud (0.006 [0.002, 0.010])** **jdate (-0.040 [-0.059, -0.021])**
	3		vine (-0.068 [-0.162, 0.026])		**cloud (0.006 [0.002, 0.010])** **jdate (-0.040 [-0.059, -0.021])**
	4			allgrass (0.024 [-0.016, 0.063])	**cloud (0.006 [0.002, 0.010])** **jdate (-0.040 [-0.059, -0.021])**

Table 3.12 (continued)

		Covariate (estimate [UCI, LCI])			
Species	**Model**	**ψ (Initial Occupancy)**	**γ (Colonization)**	**ε (Extinction)**	**p (Detection)**
Wood Thrush	5			switchgrass (-0.677 [-1.933, 0.579])	**cloud (0.006 [0.002, 0.010])** **jdate (-0.041 [-0.059, -0.021])**
	6			forbs (-0.018 [-0.052, 0.016])	**cloud (0.006 [0.002, 0.010])** **jdate (-0.041 [-0.059, -0.021])**
	7		allgrass (0.008 [-0.391, 0.536])		**cloud (0.006 [0.002, 0.010])** **jdate (-0.041 [-0.059, -0.021])**
	8		vine (-0.074 [-0.169, 0.020])	graminoid (-0.058 [-0.128, 0.011])	**cloud (0.006 [0.002, 0.010])** **jdate (-0.041 [-0.059, -0.021])**
	9			graminoid (-0.068 [-0.141, 0.005]) shrub (-0.176 [-0.418, 0.065])	**cloud (0.006 [0.002, 0.010])** **jdate (-0.041 [-0.059, -0.021])**

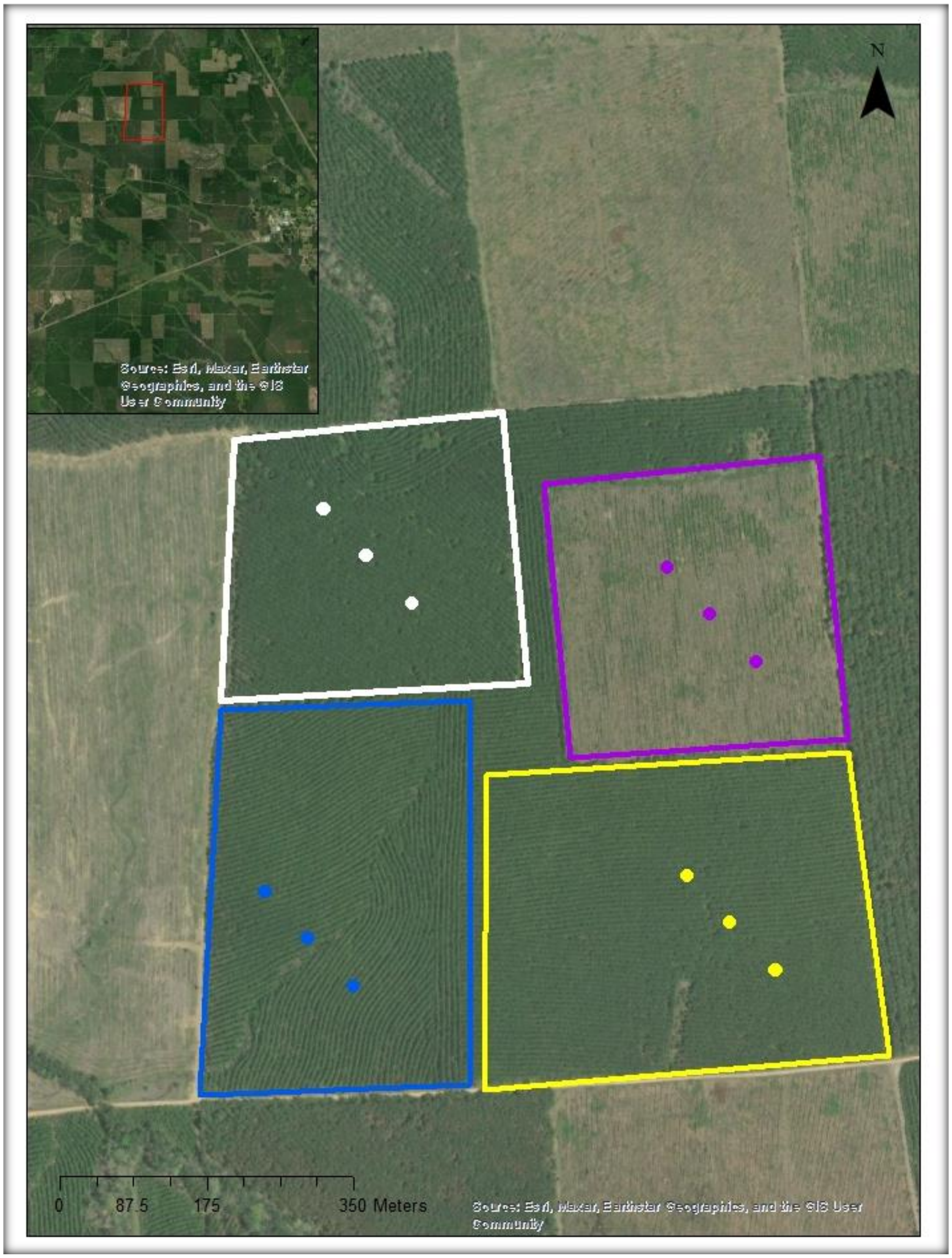

Figure 3.1 Example of avian point count survey design used during May and June 2019–2021 within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi. Points indicate survey point locations while the colored polygons designate the boundaries of each treatment: yellow - Hardwood Control; blue – Pine Control; white - Switchgrass Intercropped; purple - switchgrass monoculture (discontinued after 2011 and excluded from this study).

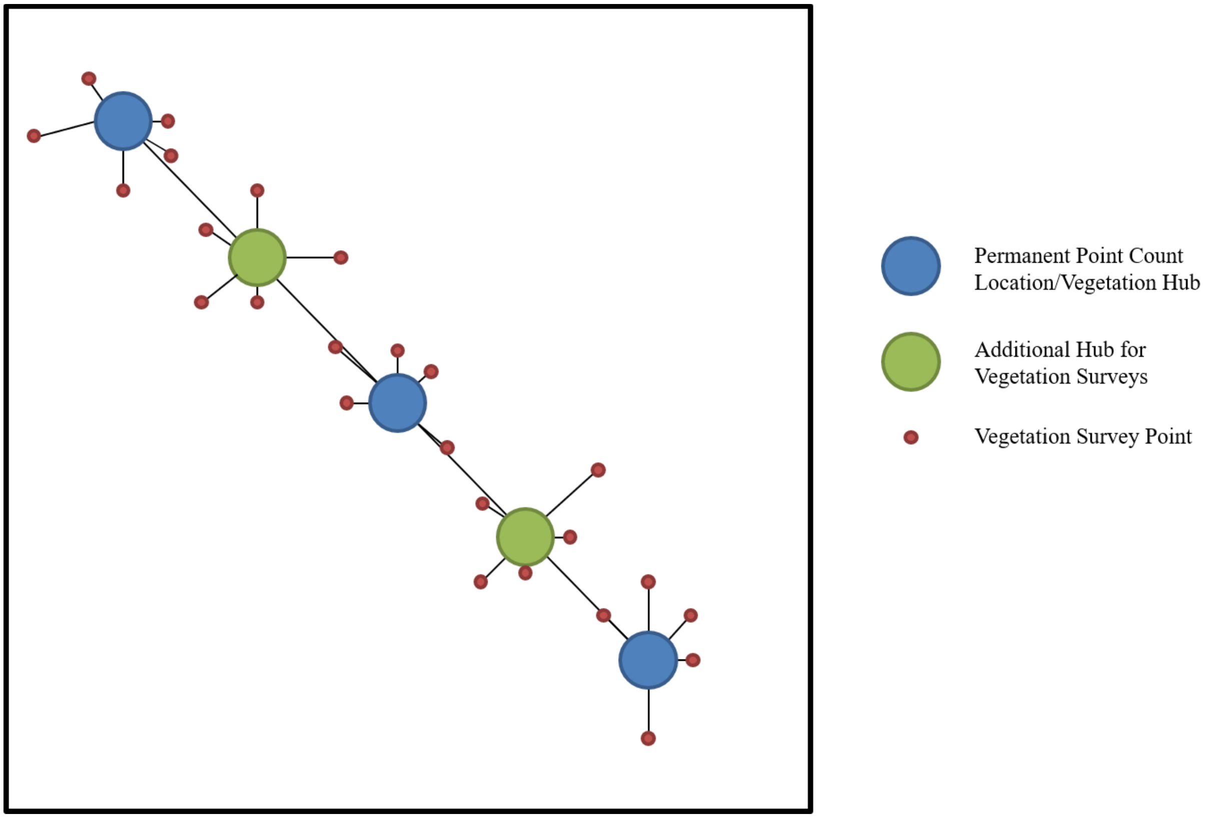

Figure 3.2 Digital representation of vegetation survey layout used within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi May–July 2019–2021. Design includes 25 random vegetation survey points in relation to permanent point count locations and two additional points equally spaced between points along the diagonal axis in each experimental study block.

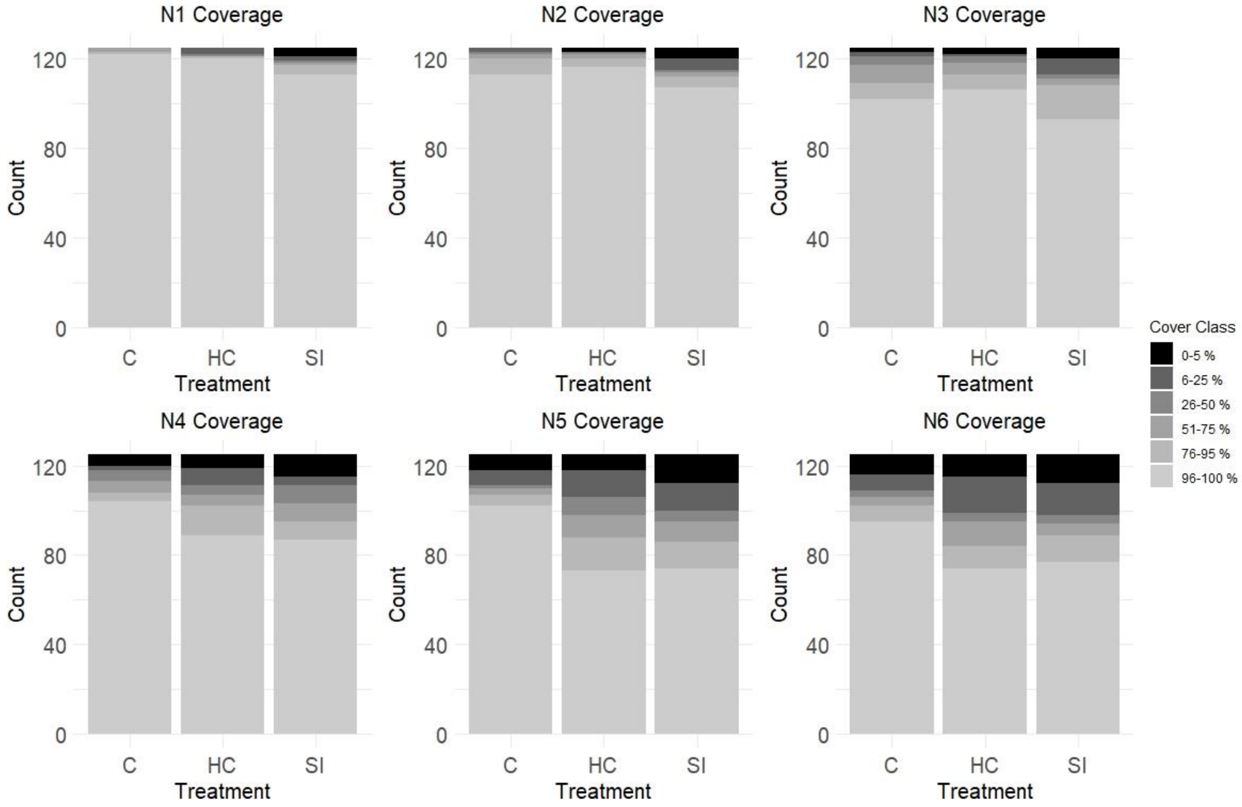

Figure 3.3 Proportional contribution of Daubenmire cover classes to all Nudds board measurements, divided into individual Nudds board sections N1 to N6, used to estimate understory density in three treatments in summer 2019 within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi.

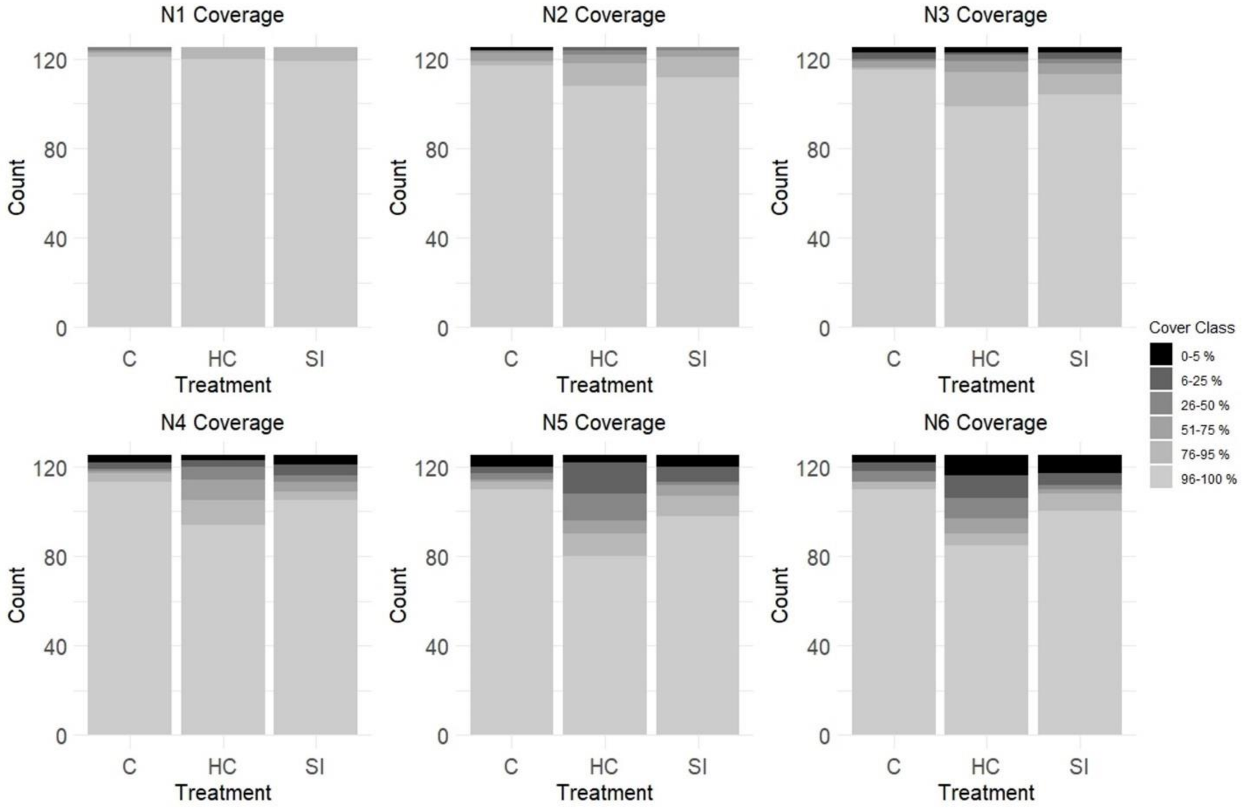

Figure 3.4 Proportional contribution of Daubenmire cover classes to all Nudds board measurements, divided into individual Nudds board sections N1 to N6, used to estimate understory density in three treatments in summer 2020 within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi.

Abbreviations: C – Pine Control, HC - Hardwood Control, SI - Switchgrass Intercropped

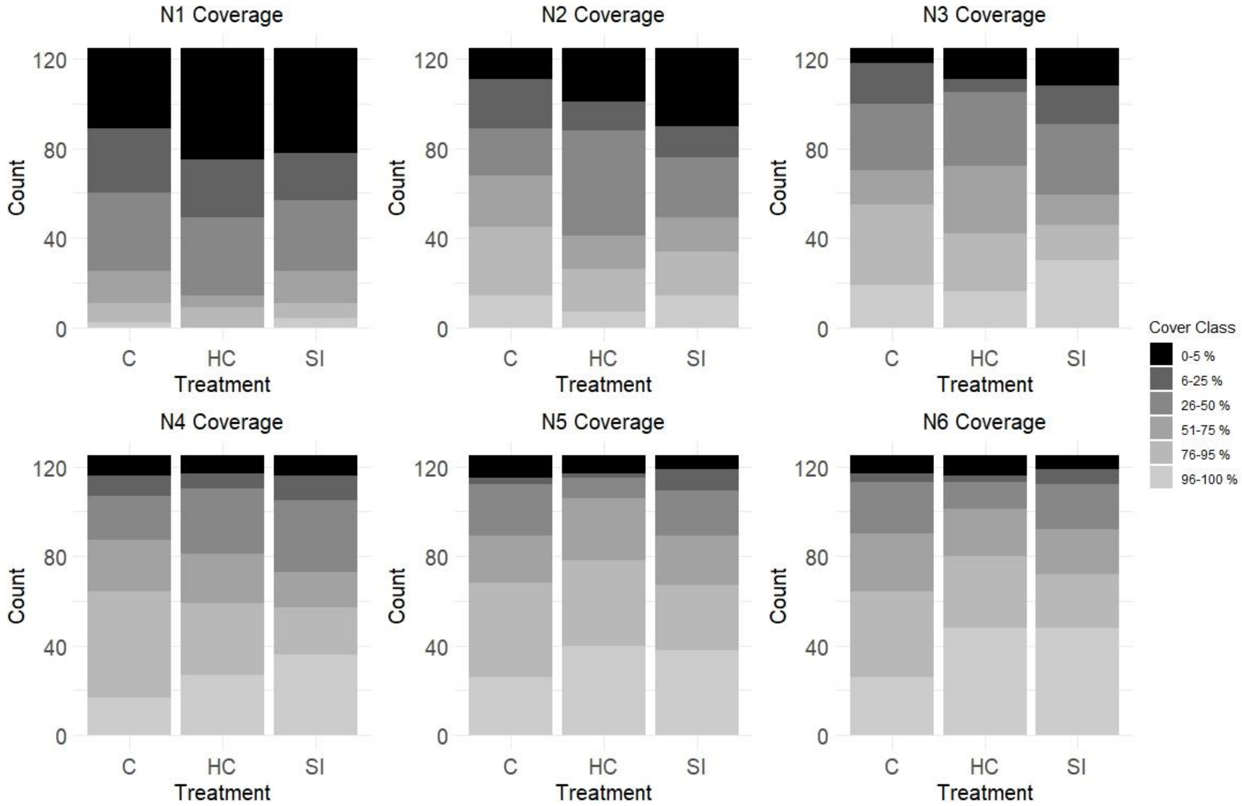

Figure 3.5 Proportional contribution of Daubenmire cover classes to all Nudds board measurements, divided into individual Nudds board sections N1 to N6, used to estimate understory density in three treatments in summer 2021 within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi.

Abbreviations: C – Pine Control, HC - Hardwood Control, SI - Switchgrass Intercropped

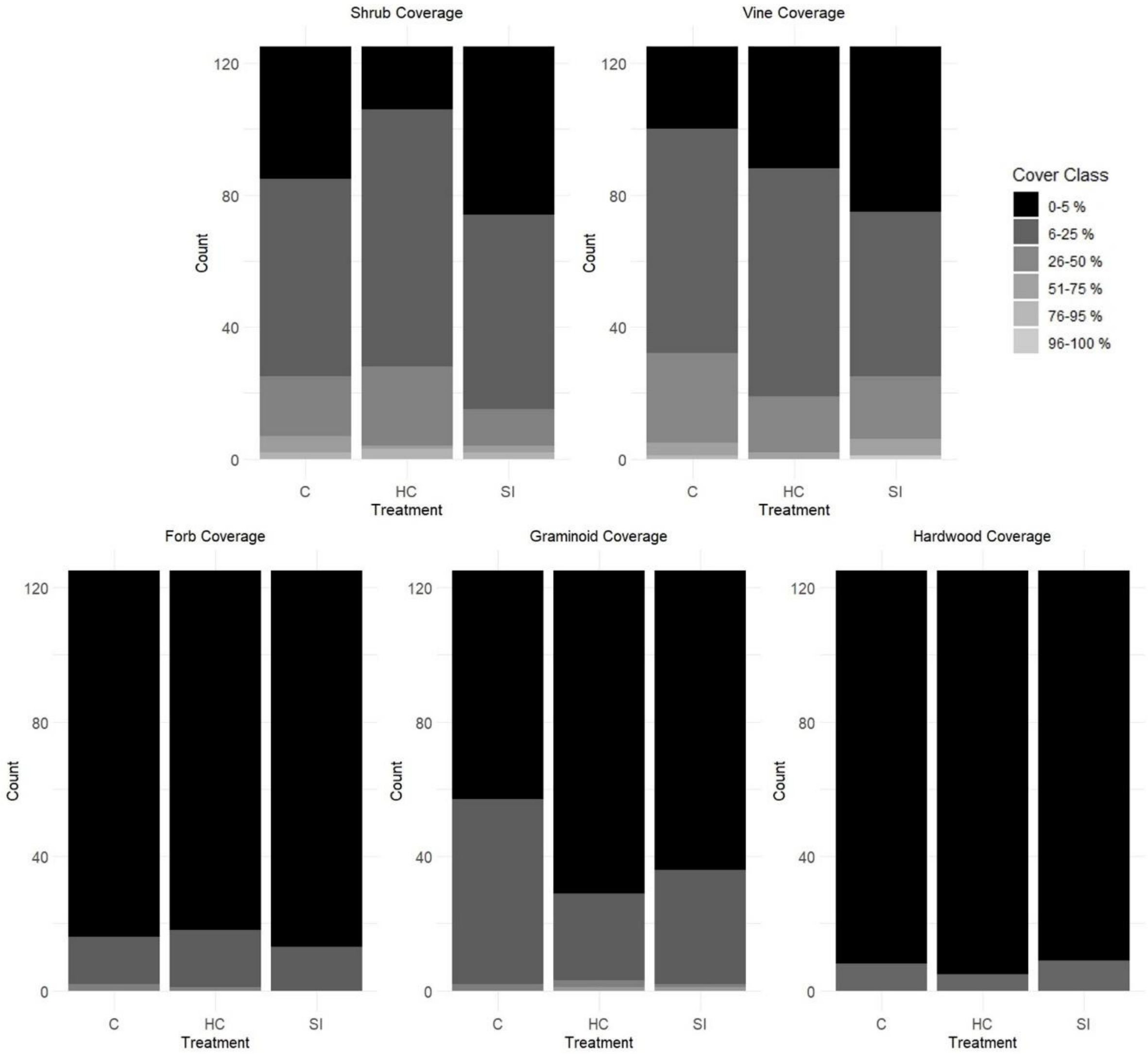

Figure 3.6 Proportional contribution of Daubenmire cover classes to all measurements of percent cover for five functional plant guilds in three treatments in summer 2019 within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi.

Abbreviations: C – Pine Control, HC - Hardwood Control, SI - Switchgrass Intercropped

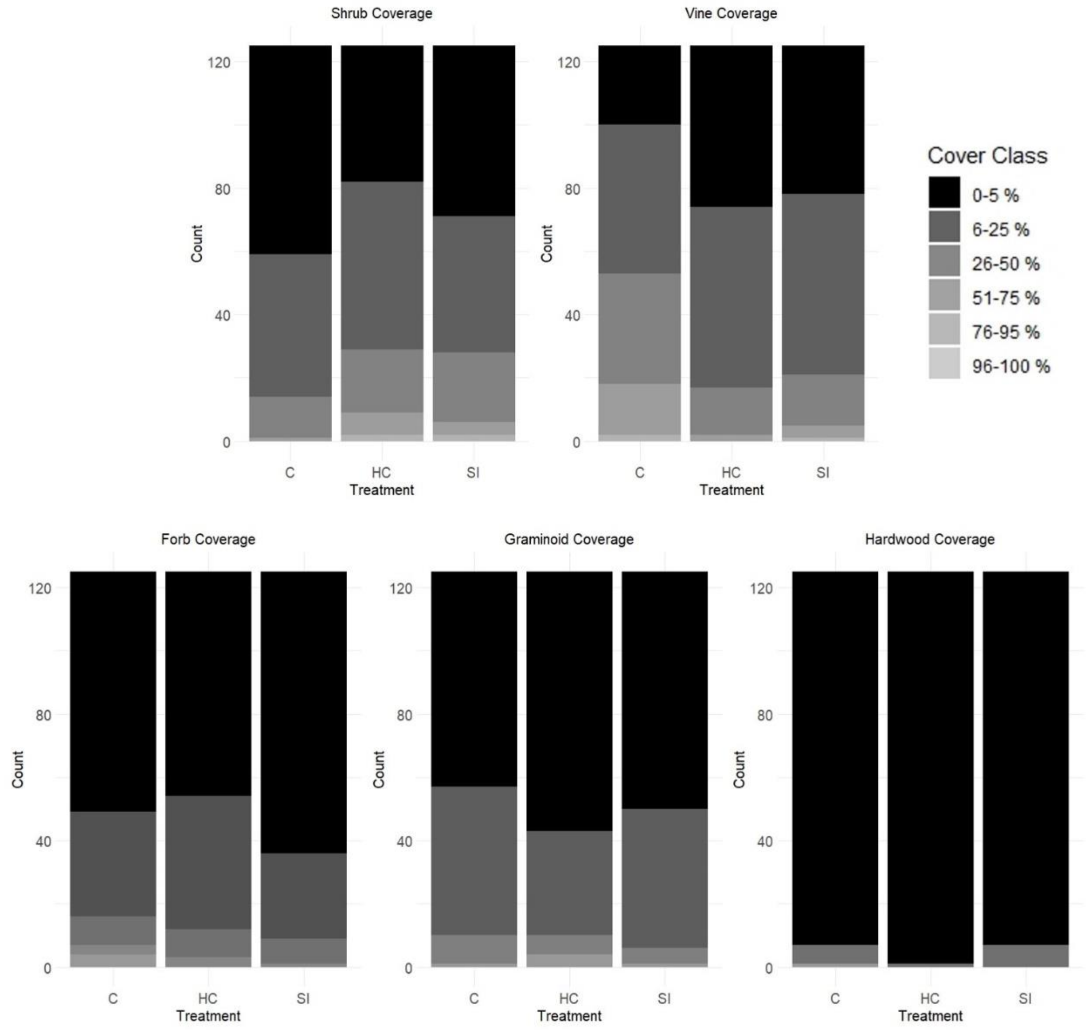

Figure 3.7 Proportional contribution of Daubenmire cover classes to all measurements of percent cover for five functional plant guilds in three treatments in summer 2020 within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi.

Abbreviations: C – Pine Control, HC - Hardwood Control, SI - Switchgrass Intercropped

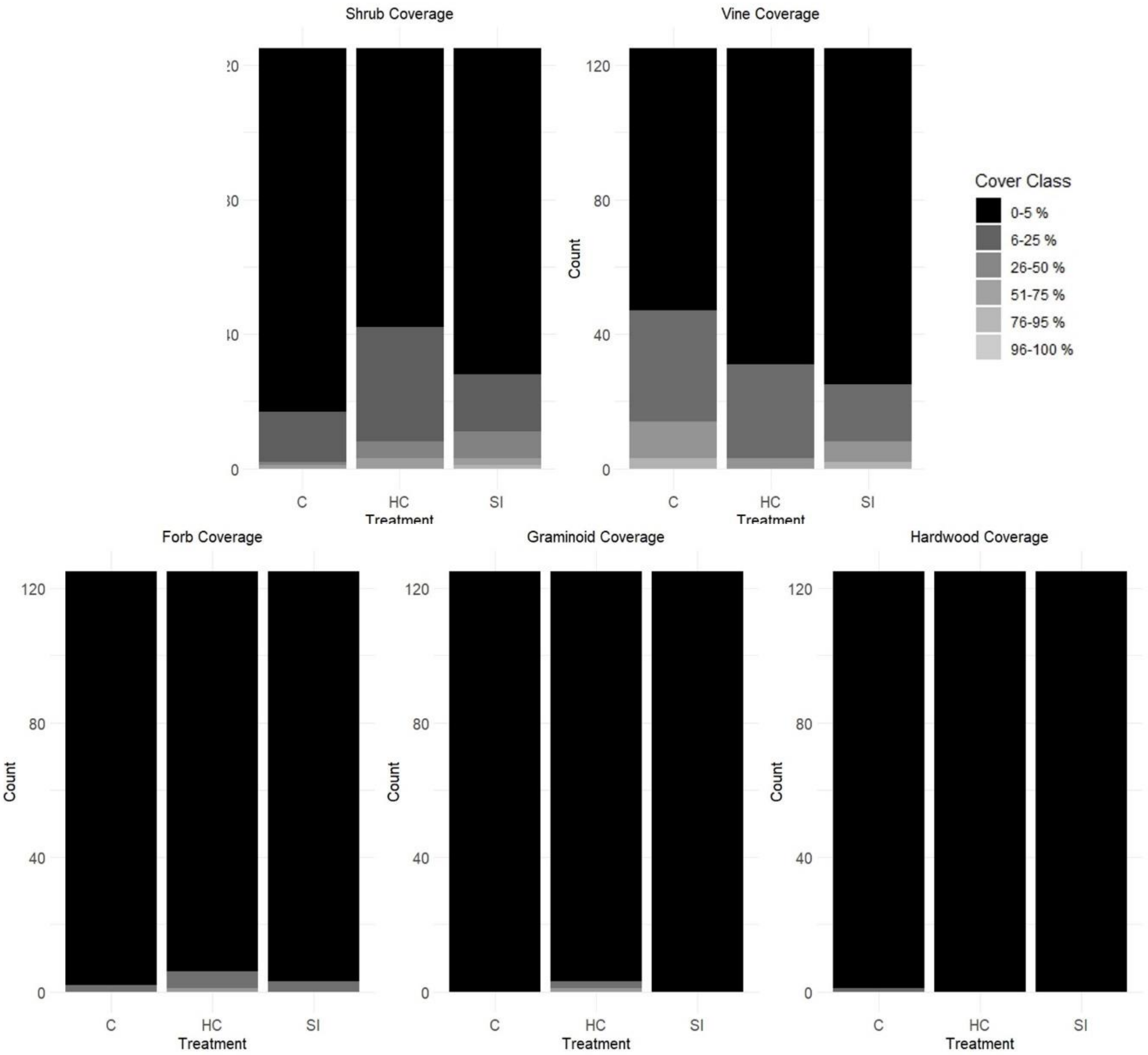

Figure 3.8 Proportional contribution of Daubenmire cover classes to all measurements of percent cover for five functional plant guilds in three treatments in summer 2021 within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi.

Abbreviations: C – Pine Control, HC - Hardwood Control, SI - Switchgrass Intercropped

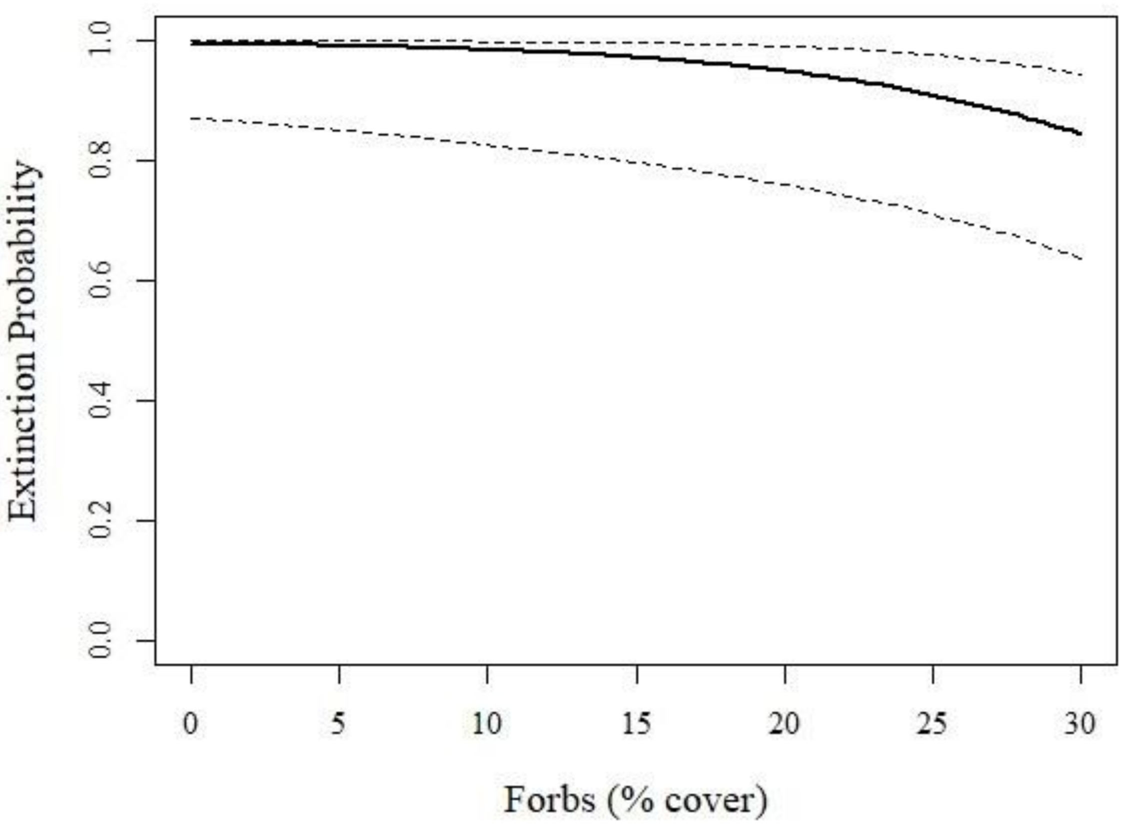

Figure 3.9 Extinction probability as a function of forb cover for Swainson's Warbler within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi from 2011–2021. Dashed lines represent 95% confidence intervals.

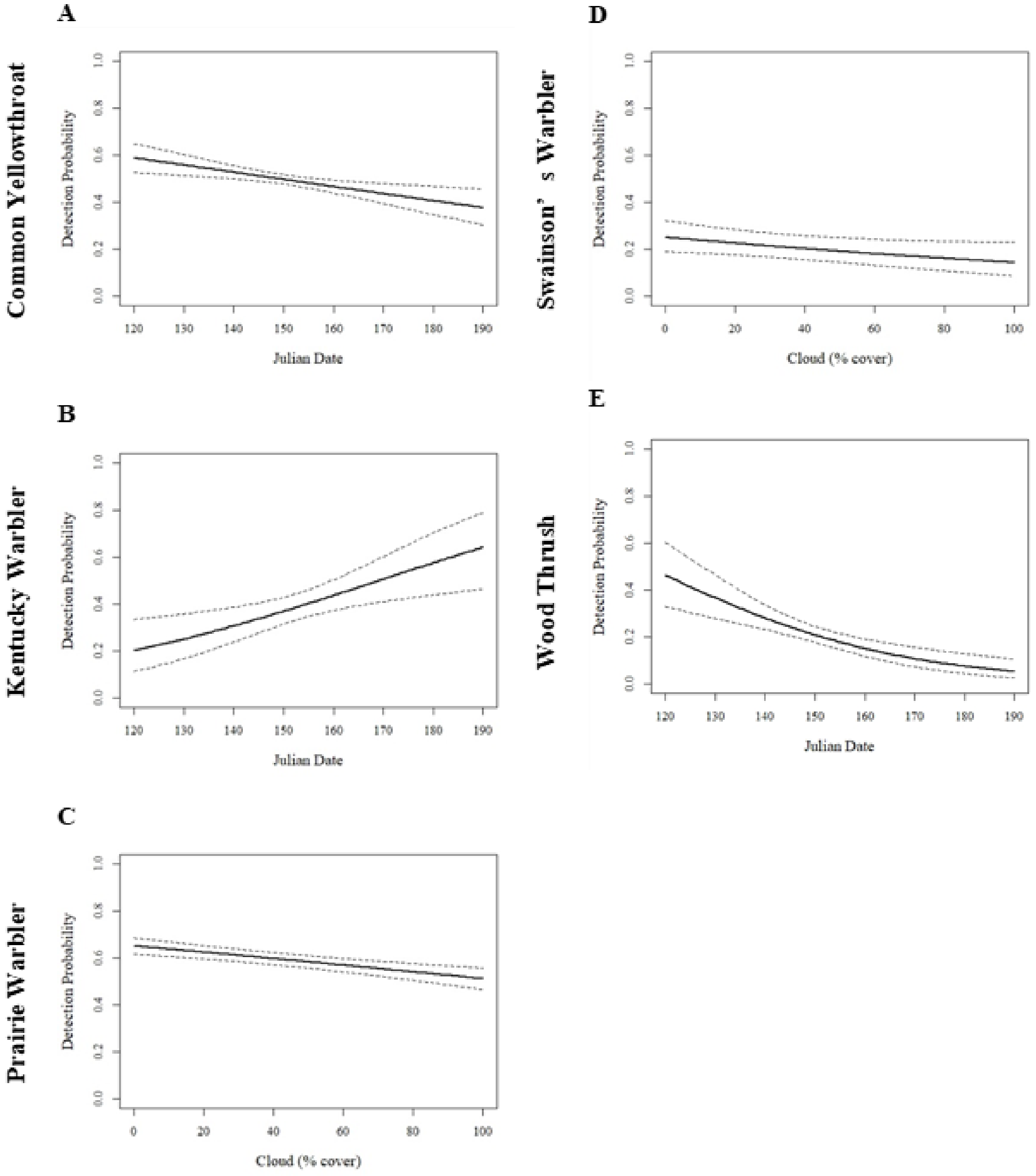

Figure 3.10 Probability of detection based on best fit detection parameters in dynamic (colonization-extinction) occupancy models for five avian species within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi from 2011–2021. Dashed lines represent 95% confidence intervals.

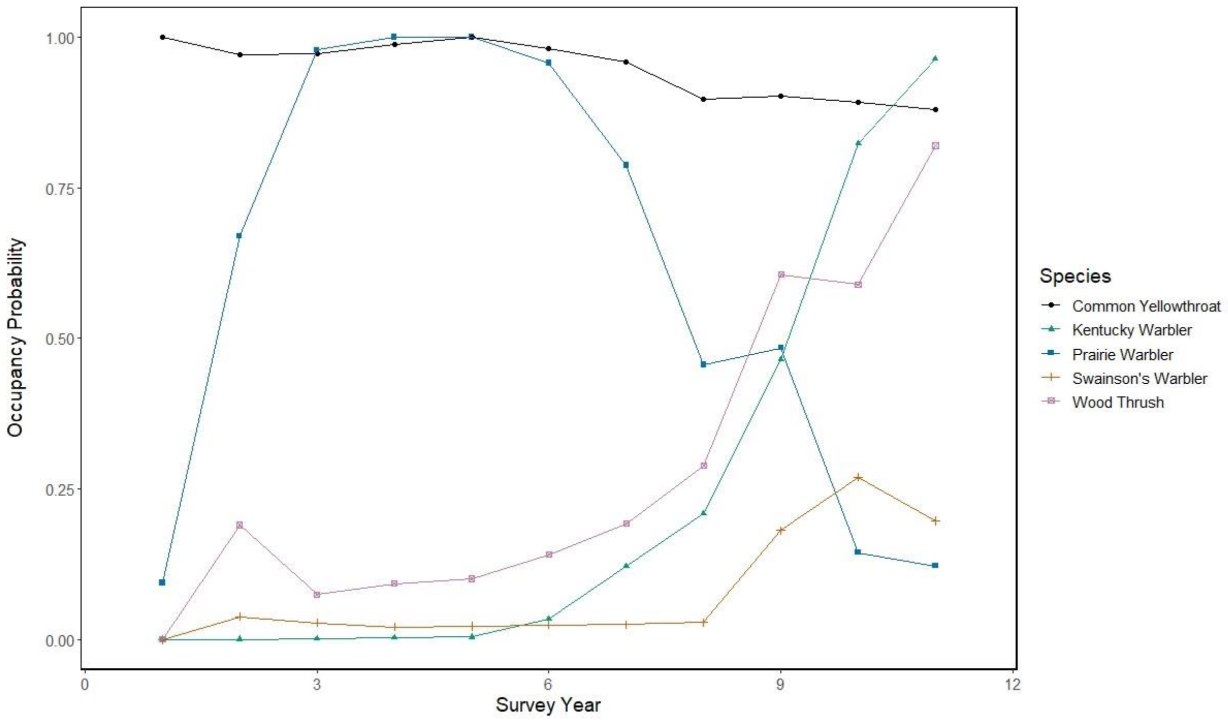

Figure 3.11 Predicted occupancy of five avian species by survey year within 5 managed loblolly pine (*Pinus taeda*) stands, starting the summer following stand initiation in 2011 and ending in summer 2021, in east-central Mississippi.

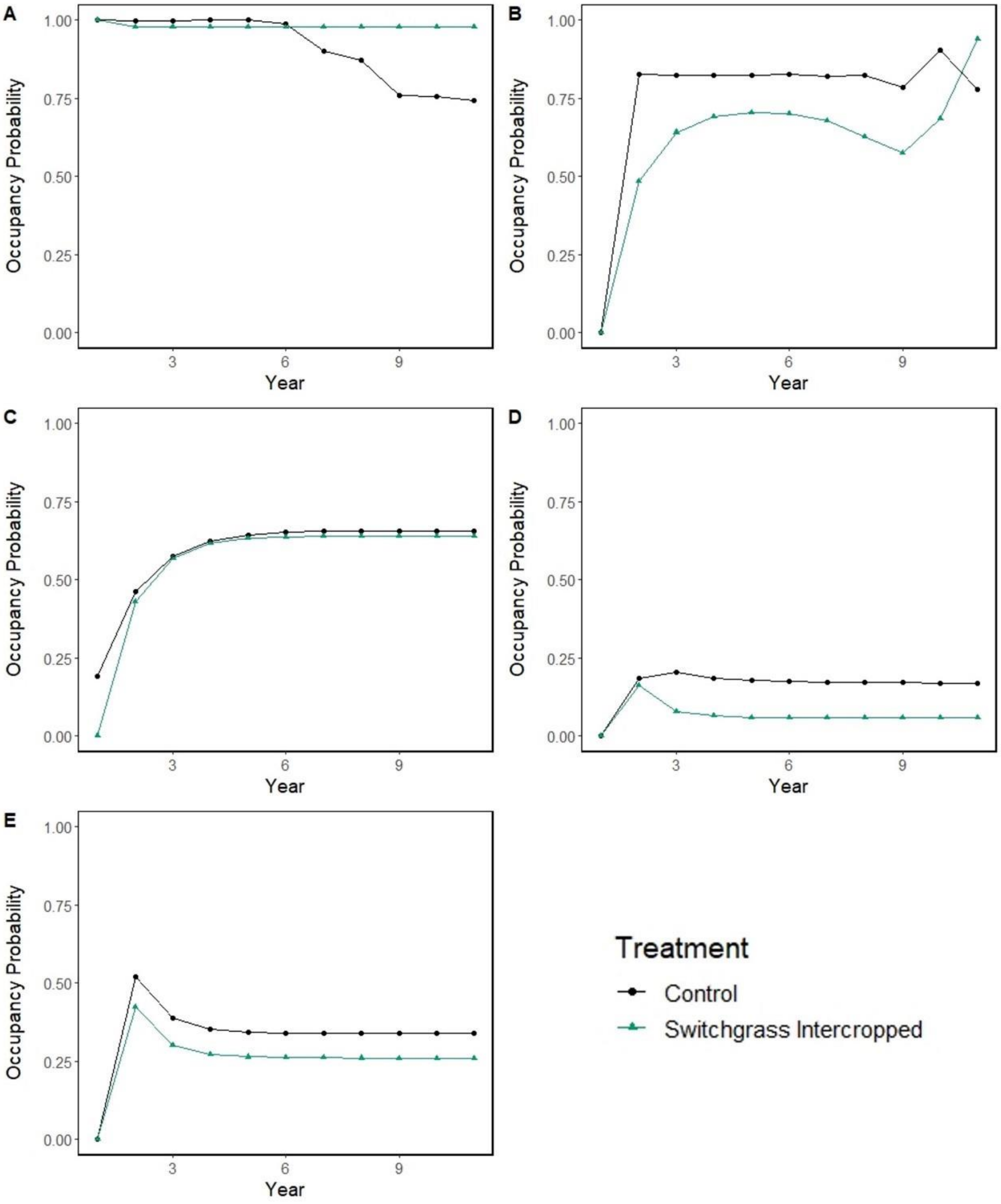

Figure 3.12 Predicted occupancy of five avian species by survey year and treatment within 5 managed loblolly pine (*Pinus taeda*) stands, starting the summer following stand initiation in 2011 and ending in summer 2021, in east-central Mississippi. A – Common Yellowthroat; B – Kentucky Warbler; C – Prairie Warbler; D – Swainson’s Warbler; E – Wood Thrush.

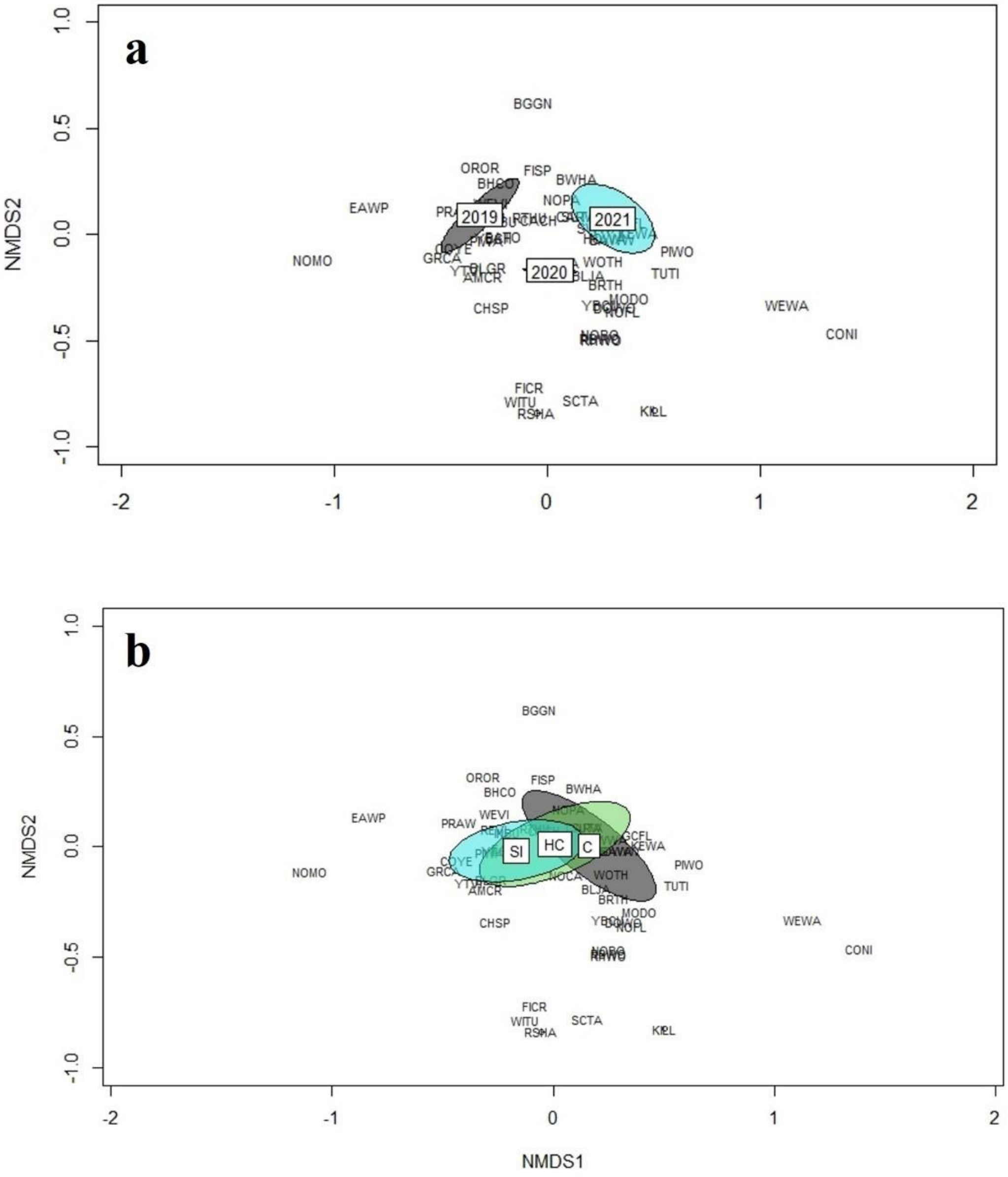

Figure 3.13 Non-metric multidimensional scaling analysis (NMDS) for avian communities within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi from 2019–2021, using species detection, (a) survey year, and (b) treatment. Bird species codes as described in Table 3.2. Treatment codes: SI – Switchgrass Intercropped, HC – Hardwood Control, C – Pine Control.

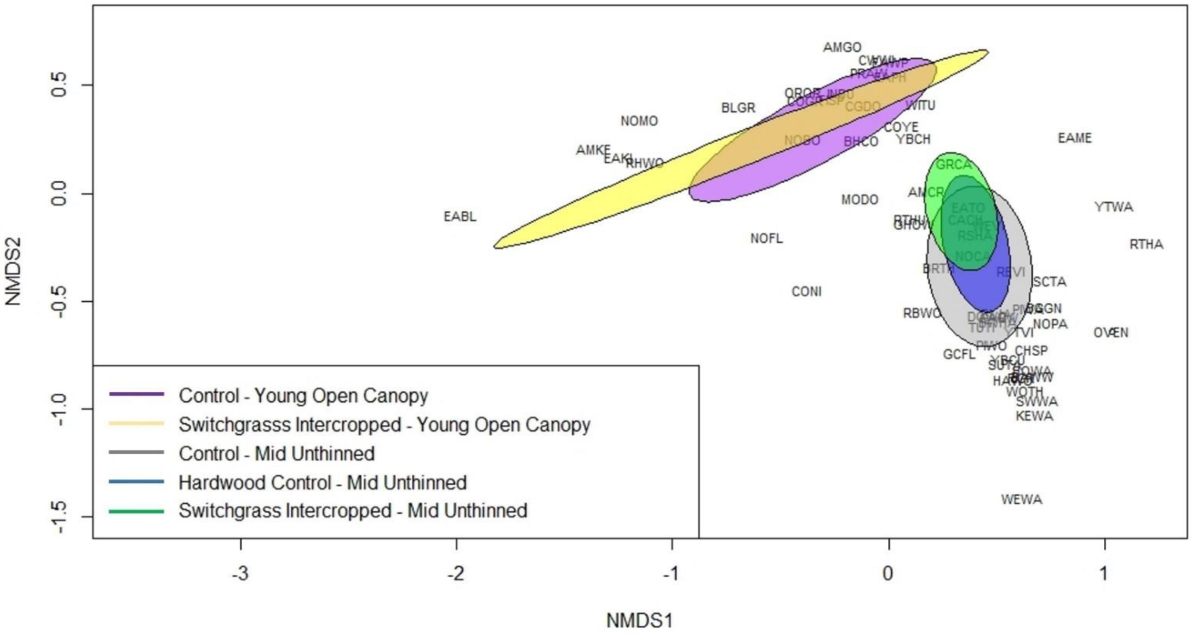

Figure 3.14 Non-metric multidimensional scaling analysis (NMDS) for avian communities within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi from 2011–2021, using species detection, treatment, and stage. Bird species codes as described in Table 3.2.

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CHAPTER IV

USE OF A MANAGED PINE FOREST BY OVER-WINTERING PASSERINES; IDENTIFYING AVIAN SPECIES DIVERSITY, PARASITE PREVALENCE, AND ISOTOPIC NUTRIENT ASSIMILATION IN STANDS OF DIFFERENT AGES

Historically, most avian research has focused on breeding and migratory efforts. However, over-wintering studies are increasingly recognized as important in understanding the suite of factors that affect the full life cycle of migratory birds (Burton et al. 2006, Norris and Marra 2007, Faaborg et al. 2010, Dybala et al. 2015). After severe population declines of many forest species in the 1960s and 1970s, loss of ecosystem conditions supporting species during winter was identified as one of the main contributing factors (Askins 1993). Temperate forests in the winter contain markedly lower abundances and diversity of avian species when compared to the breeding season in the same location (Childers et al. 1986). While extensive research has been conducted throughout much of the United States during the breeding season, minimal information exists about how forest management practices affect birds outside of their breeding seasons, especially during winter (Hanberry et al. 2013). As managed forests are subjected to a continual rotation of planting and harvesting, plus intermediate treatments (e.g., thinning, herbicide use, and prescribed fire), questions remain about how individual birds use each managed forest stage during winter. Equally important is the question of why some species, and consequently certain individuals, reside in a particular area during times of differing environmental conditions (e.g., differences in food resources, thermal refugia).

As during the summer months, avian species composition during winter changes over time as forest stands age (Childers et al. 1986). Less intensive pine establishment methods have been shown to result in higher levels of richness and abundance of winter avian species shortly after stand initiation (Darden 1980, Hanberry et al. 2013). However, abundances and diversity of avian communities within variously aged stands may not exhibit marked differences, as documented in multiple structurally different ecosystems (Noble and Hamilton 1976, Conner et al. 1979, Childers et al. 1986, Grodsky et al. 2016).

Managed pine (*Pinus* spp.) forests in the southern United States have displayed high avian abundance levels during winter, particularly for species such as Savannah (*Passerculus sandwichensis*), Song (*Melospiza melodia*), and Swamp (*Melospiza georgiana*) Sparrows (Grodsky et al. 2016). The availability of structured vegetation and downed woody debris within these systems may influence the local abundances of avian species within different forest patches (Rost et al. 2010, Grodsky et al. 2016). Within recently harvested forest stands, remaining woody debris may play an ecological role complementary to vegetation for wintering avian populations (Grodsky et al. 2016). During the winter, individual birds may adjust their space use depending on food availability, and species commonly associated with early successional shrub vegetation are widely distributed across the available forest habitat (Schmiegelow and Monkkonen 2002, Hamel 2003). Furthermore, avian populations may display different types of seasonal movements within the areas where they over-winter. These movements may range from a wandering food searching pattern, varying in extent and intensity and exhibiting patterns related to the optimal foraging theory, to fixed movements largely independent of the environment and often classified as transient in nature (Helms and Drury, Jr. 1960). In areas of higher edge densities, including in managed forest landscapes, decreases in tall vegetation can cause decreases in availability of thermal refugia and lead to altered foraging and roosting behaviors (Kwit et al. 2004, Elsen et al. 2020, Howell et al. 2021). Therefore, species can rely on a mixture of cover types during winter, provided by an assortment of stand ages, structures, and compositions across a managed pine landscape that can enhance a landscape's salience to birds.

One method to identify use of cover types via nutrient assimilation is through using stable isotope analysis, comparing blood isotope from wildlife species, and individuals, to carbon and nitrogen ratios in sampled vegetation. Stable isotope analysis is used to investigate migratory pathways in many species but is more recently being applied to infer how organisms use resources within their habitats (Inger and Bearhop 2008, Evans et al. 2012, Fuller-Morris et al. 2020). Carbon assimilation in avian blood samples can be compared to vegetation samples, specifically C3 and C4 plant species to identify sources of primary productivity assimilated by the organism sampled. For instance, individuals with more negative carbon values likely rely heavily on wooded vegetation while less negative values indicate a reliance on grassland environments (Smith and Epstein 1971, Kelly 2000, Evans et al. 2012). Although most plants are C3 photosynthesizers, many plants in hot and dry environments rely on the C4 photosynthetic system to save energy. In southern managed forest landscapes, examples of C3 plants include pines (*Pinus* spp.) and many shrub species, while C4 plants such as switchgrass (*Panicum virgatum*), maize (*Zea mays*), and sorghum (*Sorghum* spp.), are often found in agricultural environments. These plant species have divergent photosynthetic pathways resulting in different isotope values (Smith and Epstein 1971, Kelly 2000).

Information on trophic hierarchies can be acquired through measurement of nitrogen isotope ratios, with ratios exhibiting a stepwise enrichment through food chains (Ambrose and DeNiro 1987, Hobson and Welch 1992, Hobson and Clark 1992). Isotopic turnover rates vary by animal mass and specific tissue type, with larger individuals retaining previously assimilated isotopic values for longer periods of time; however, isotopic blood turnover rates for many avian species averages 3–6 days (Hobson and Clark 1993, Vander Zanden et al. 2015). Carbon is used for most isotopic turnover studies, while nitrogen is more commonly used to measure turnover of proteinaceous tissues, which, on average, have a longer turnover time (Bauchinger and McWilliams 2009, Carter et al. 2019). But carbon and nitrogen turnover rates can have opposing responses to ecological stresses (Colborne et al. 2017). Examining tissue nitrogen, while becoming more common in ecological studies of dietary nutrient assimilation, should be approached cautiously given the potential influence of physiological effects instead of diet (Hobson and Clark 1993). Short term, or even seasonal, dietary analysis can thus be investigated with this method, as blood isotope values, compared to muscle and other proteinaceous tissues, change rapidly enough to be sensitive to small changes in prey consumption.

Over-wintering migratory and resident avian species are affected by multiple conditions, such as decreasing temperatures, potential limits in food sources and food stores, and increased competition for available food resources. One often overlooked aspect of over-wintering avian condition lies in blood parasitic infections, and the potential affects parasites can have on vulnerable avian populations. Blood parasitic infections are chronic, but the ability to detect prevalence through common sampling techniques such as blood smears varies throughout the year due to climatic and physiological stressors and limitations in sampling techniques (Goater and Holmes 1997, Fallon and Ricklefs 2008). In wild avian populations, effects of blood parasites may also not be detected because of varying parasitic life cycles (Pérez-Rodríguez et al. 2015). While parasitemia is rarely fatal, birds with high levels of parasites are also thought to be less likely to be caught and sampled, generally thought to be a consequence of restricted movements and thus a decreased likelihood of being caught by common trapping techniques (Valkiūnas 2005).

Parasitic infections can be found throughout the year because transmission can occur continuously in both breeding and wintering areas, with *Haemoproteus* infections remaining for months after initial infection (Valkiūnas 2005, Soares et al. 2020). But infection prevalence is thought to decrease temporally during the winter months (Dunn et al. 2014). Similarly, many of the vectors for blood parasites are dormant during the winter months, thus reducing transmissions rates and new infections (Dunn et al. 2013).

Winter conditions of cooler temperatures and food restrictions, relative to the vernal period, may lead to increases in stress, including increasing corticosterone levels, and stress-induced decreases in immunity that may trigger parasite relapse (Barrow 1963, Valkiūnas 2005, Dunn et al. 2014). With the potential for multiple stress inducing factors to have synergistic effects, it can be difficult to attribute any effects on the host's fitness to parasitic infections, even though most wild birds likely contain chronic parasitic infections (Clinchy et al. 2004, Valkiūnas 2005, Dunn et al. 2014). Studies investigating parasitism during the non-breeding season are rare, yet there are potentially important implications of winter infections, when elevated levels of stress, with the potential to exacerbate parasitic infections, might be present due to decreases in food availability and colder and more extreme weather conditions (Allander and Sundberg 1997, Romera et al. 2000, Kitaysky et al. 2001, Valkiūnas 2005, Dunn et al. 2014). During winter months when environmental stresses include low temperatures, increased flocking behavior, and requirements for scarce food sources, bird populations may be affected by the additional pressures created by parasitic infections (Valkiūnas 2005).

Given the lack of understanding of over-wintering avian community dynamics, particularly within working forests, research was needed to address knowledge gaps regarding resource use and individual condition. Specifically, there was a need to investigate if early successional and woodland bird species use different forest resources during winter when compared to their known breeding habitat associations. Therefore, I estimated over-wintering avian richness by using capture-based methods and incidental reports with four different forest stands, a Young Open Canopy, Mid Unthinned, and two Late Stage stands. I examined how capture rates were influenced by forest stand stage, temperature, and time of year. Additionally, I explored isotopic nutrient assimilation as a proxy for investigating dietary intake and investigated over-wintering external and internal parasite prevalence in captured individuals.

I predicted avian species richness would be greatest in the Young Open Canopy stand, a stand where most early successional avian species should be found regardless of season. This stand age is known to support many high-quality food sources during summer months (Swanson et al. 2011) and given the decreasing amount of understory vegetation in aging forest stands within our study area, I suspected this could be true of the winter months as well. I predicted capture rate would be greatest in November and March, when many birds are ending and starting their migratory periods, respectively. However, I also predicted capture rates would increase as temperature decreased, as individuals would spend less time sedentary and more time actively searching for food. Given the propensity of individuals to use a variety of resources during the winter months, I expected isotopic nutrient assimilation, specifically carbon ($^{13/12}$C, hereinafter δ^{13}C) and nitrogen ($^{15/14}$N, hereinafter δ^{15}N), to be similar among all species, with only species that are known to only use open conditions to have significantly different values. Finally, I anticipated over-wintering parasite prevalence to be low given prior research (e.g., Cosgrove et al. 2008, Soares et al. 2019.

Methods

Study Site

See Chapter 2 of this dissertation for full description of study area.

Experimental Design

I chose three managed loblolly pine (*Pinus taeda*) stands from within the larger forested landscape, including:

1. One young, open canopy stand (hereinafter, Young Open Canopy), established in 2016, that was managed following standard Weyerhaeuser silvicultural management practices for site preparation and loblolly management, including herbaceous weed control and fertilizer application. The stand contained 916 trees per hectare and was 57.43 hectares in size.
2. One mid-age, unthinned stand (hereinafter, Mid Unthinned), established in 2010, was managed for loblolly pine and from 2011–2015 intercropped with switchgrass. The stand contained 1,319 trees per hectare and was 8.25 hectares in size.
3. A mix of two older stands (hereinafter, Late Stage), one established in 1995 and one in 2005, that were managed under standard Weyerhaeuser protocols. The older stand was 30.28 hectares in size and was mechanically thinned to 427 trees per hectare 11 years prior to the start of the study. The younger stand contained 1,064 trees per hectare and was 15.20 hectares in size. Both stands were fertilized in 2009.

I chose specific stands based on age and vegetation structure (dense grass and shrub mixed with pine in the Young Open Canopy stand; patches of dense shrub and open areas in the Mid Unthinned stand, with a closing canopy; and little understory vegetation with a closing canopy in the Late Stage stands), ease of access from the main roads, and proximity to a chosen central location that was used as a banding station.

I placed 12 m Avinet 30 mm polyester mist nets in all three stands to capture the highest avian diversity. In total, I set up 15 nets during each survey day, five in each stand type (Figure 4.1). I placed nets 50 to 100 m apart to maintain sampling independence between nets (Desante et al. 2019). In some cases, nets were arranged closer than the ideal placement due to thickness of the surrounding vegetation and ease of access. I created a banding station on the edge of the survey site and released captured individuals from this location to minimize handling time. I checked nets on a rotation throughout the day, no more than one hour between net runs.

Following protocols established in MoSI (Monitoreo de Sobrevivencia Invernal - Monitoring Neotropical Migrants in Winter), I conducted sampling on three consecutive days, repeated monthly for five winter months, November through March 2019–2022 (DeSante et al. 2009). I operated nets for 6–8 hours on each day of operation. In the event of inclement weather, I only opened nets for as many hours as conditions allowed, or a banding day was rescheduled for the next appropriate day when possible. During each banding day, I kept a list of all avian species seen and heard while the nets were open. The survey list aimed to identify any species that were not captured in the mist nets. For each species detection, I recorded approximate location and pine stand stage. I recorded net opening and closing time, opening and closing temperature, cloud cover, and wind speed.

Bird Banding and Blood Isotope Samples

I identified each captured individual to species, banded each with a uniquely numbered USGS leg band, aged, identified the sex of the bird when possible, weighed to obtain mass (g), measured unflattened wing chord (mm), examined for fat storage, and checked for external parasites (USGS permit #23835-F). I also recorded the net number from which each bird was captured and the subsequent stand stage, and time of capture.

For each individual, I collected a small blood sample from the brachial vein, using a sterile 27-gauge needle. I collected blood samples in non-heparinized capillary tubes (volume = 74 µl). I obtained less than 1 % body weight recommended as a best practice (Fair et al. 2010). I created a blood smear with a small amount of the collected blood, and the remaining blood was transferred to a 2 mL cryogenic externally or internally threaded storage vial.

I collected small plant samples of what was determined to be representative of the study area at the beginning of the final season. I collected leaves from American Sweetgum [*Liquidambar styraciflua*], Chinese privet [*Ligustrum sinense*], Sumac [*Rhus* spp.], Oak [*Quercus* spp.], and *Rubus* spp., blades from switchgrass [*Panicum virgatum*], and needles from loblolly pine [*Pinus taeda*]). After collection, I stored vegetation and blood samples at -80º C until the end of the season, when all samples were pressure dried and shipped for analysis. I sent samples to The Center for Applied Isotope Studies at the University of Georgia (Athens, Georgia) for analysis of total C, total N, and isotopes of carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$), following methods used in previous studies at the same field site (Fuller-Morris 2018).

Parasite Sampling

I surveyed each captured individual for external and blood-borne parasites. I visually inspected each bird for ectoparasites, commonly feather mites (superorder Acariformes) and ticks (suborder Ixodida). I scanned multiple areas of the bird, including the remiges, around the auriculars, eyes, and lores, around the legs, and around the cloaca. Each location served as a different site, and consequently, an individual survey. In addition, I asked at least one additional person to copy my visual surveys to increase the likelihood of detection and create survey replicates. When I found feather mites on the remiges, mite load was categorized as none, low (defined as only a few feather mites seen, and not on every flight feather), medium (defined as observing an average of one feather mite on most to all flight feathers), and high (defined as more than one feather mite on most or all flight feathers).

During the blood collection process, I created a blood smear from blood collected from each bird. Smears were air-dried in the field and stained with modified Giemsa stain (Electron Microscopy Services Differential Quick Stain Kit) upon return to the laboratory. The stain contained Azure, methylene blue, and eosin dyes, acidic and basic components, that stain the nuclei and cytoplasm in erythrocytes, leucocytes, platelets, and parasites. Stained slides were stored in a sealed slide box until examination, to prevent debris contamination and damage. The stained slides were examined at 1,000× magnification under oil immersion. Each blood smear was examined for blood-borne parasites including but not limited to Haemosporida (*Haemoproteus* spp., *Leucocytozoon* spp., and *Plasmodium* spp.) and the genus *Trypanosoma*. Each smear was checked by two observers to create replicates and maximize the likelihood of detection.

Statistical Analysis

I determined capture rate for each month, winter session (2019–2020, 2020–2021, and 2021–2022), month and stand, stand and session, stand total, and total across all sessions. Capture rate is determined by dividing the number of successful captures by mist-net hours. I used capture rate instead of capture numbers in all models to normalize unequal sampling efforts between stands, months, and sessions. For all models, results were considered statistically significant at $\alpha < 0.05$.

All analysis was completed in R version 4.0.3 (R Core Team 2020). I used linear models (LM) to evaluate the effects of session, capture location, capture month, species known habitat association (as assigned by the North American Breeding Bird Surveys [Sauer et al. 2017]), average opening temperature (AvgOTemp), and average closing temperature (AvgCTemp) on capture rate. I created models with single covariates, followed by models of all possible combinations of covariates. I used *modsel* in *unmarked* (Fiske and Chandler 2011) to create a model selection table and ranked models by second order Akaike Information Criterion (AICc; AIC corrected for small sample sizes) and log likelihood (Akaike 1973; Table 4.1). I assessed goodness-of-fit of models using a ratio of residual deviance to null deviance, which is used to create an R-squared value. I assessed models for homogeneity of variance by plotting the model residuals and creating a Normal Q-Q plot and checked for normality of data using a Shapiro-Wilk normality test (R Core Team 2020).

I used LMs with δ^{13}C and δ^{15}N as the response variable to determine if differences in dietary isotopic values could be attributed to predictor variables including species, species habitat guild, and capture location. Following methods used for capture rate analysis, I created models with single covariates, followed by models of all possible combinations of covariates. I used *modsel* in *unmarked* (Fiske and Chandler 2011) to create a model selection table (Table 4.2), and rank models by AICc and log likelihood (Akaike 1973). I assessed goodness I used the *SIBER* package to plot δ^{13}C and δ^{15}.

Results

Species Richness and Captures

During the 3 winter sessions (November 2019–March 2020, November 2020–March 2021, and November 2021–March 2022), I banded 110 individuals in 131 total captures, representing 15 species (Table 4.3). I also observed (auditory or visual) 29 species (Table 4.4) of which 18 were not captured in mist nets. The Young Open Canopy provided the greatest number of captures (44 %), whereas only 19 % of all captures were in the late rotation stand across all years. November produced the highest number of captures across all months and years, with 31 % of all captures, followed by January with 29 % of all captures (Table 4.5). Multiple banding days in each season were canceled due to potential weather or nearby forest management activities. Seven days were canceled in winter 2019–2020, two days in winter 2020–2021, and five days in winter 2021–2022.

I recaptured two Carolina Wrens (*Thryothorus ludovicianus*), one Eastern Phoebe (*Sayornis phoebe*), one Golden-crowned Kinglet (*Regulus satrapa*), five Hermit Thrushes (*Catharus guttatus*), nine Ruby-crowned Kinglets (*Regulus calendula*), and one Song Sparrow within the three winters sessions (Table 4.6). The first recaptures, two Ruby-crowned Kinglets, occurred during the 2nd month of banding sessions (December 2019), with the last recapture, also a Ruby-crowned Kinglet, in March 2022. Of these, two individuals were recaptured more than once. Of all the recaptures, 81% were recaptured in the same stand as their original capture location, whereas five individuals were recaptured in the same net. Forty-seven percent of all recaptures occurred in the Mid Unthinned stand, while 71% of all recaptures occurred during the 2020–2021 season. Mean distance between recaptures was 94.84 m (0 – 289.93 m).

Capture Rate

Capture rates were greatest in the Young Open Canopy stand and least in the Late Stage stands (Table 4.7). I found three competing models for predicting capture rate, though only one did not violate normality assumptions. The top model included capture location and average closing temperature at time of net closure (Shapiro-Wilk $P = 0.033$, $R^2 = 0.26$), followed by capture location, average closing temperature at time of net closure, and session (Shapiro-Wilk $P = 0.159$, $R^2 = 0.37$) and capture location (Shapiro-Wilk $P = 0.007$, $R^2 = 0.14$). Capture rates varied significantly by location, closing temperature, and session. As closing temperature increased, capture rates decreased (P < 0.05; Table 4.8), capture rates were significantly higher in the Young Open Canopy stand across all sessions (P < 0.05), and there were statistically lower capture rates during the third session (2021–2022; P < 0.05).

Blood Isotope

The analysis of the first 26 isotope samples, collected November 2019 through December 2020, revealed that there was a significant relationship between species habitat guild and isotopic carbon (Table 4.9). Mean $\delta^{13}C$ in woodland species was -25.54 (SD = 1.22) whereas mean $\delta^{13}C$ in early successional/scrub species was -19.92 (SD = 4.02; Table 4.10). Species classified as early successional species had significantly greater (less negative) $\delta^{13}C$ values (P < 0.05; Table 4.9). Among species with at least two samples from different individuals, I found that both sparrow species (Song and Swamp) had significantly different $\delta^{13}C$ values than the remaining species (Figure 4.2). Both these species were only captured in the Young Open Canopy stand (Figure 4.3).

Woodland and early successional/scrub species had different mean $\delta^{15}N$ values (4.86 ± 1.45 and 4.58 ± 0.76, respectively; Table 4.10). With $\delta^{15}N$ as the response variable, using species as the predictor produced significant results (Table 4.9). Early successional species had higher $\delta^{15}N$ values when compared to woodland species (P < 0.05). Among species with at least two samples from different individuals, only Swamp Sparrow had significantly different $\delta^{15}N$ values relative to the other species sampled (Figure 4.2).

Parasite Surveys

During the three winter sessions, I made blood smears from 85 individuals. Of those 85 samples, I found 4 samples containing bloodborne parasites. This resulted in a confirmed bloodborne parasite rate of 5 %. Three individuals, two Ruby-crowned Kinglets and one Eastern Towhee (*Pipilo erythrophthalmus*) had *Haemoproteus* spp. while one Golden-crowned Kinglet as infected with *Trypanosoma* spp. (Table 4.11). Both parasitic genera are protozoa that infect the erythrocytes and have previously been documented in Passerines found around the world (Figure 4.4; Clark et al. 2014).

A visual check of all captured individuals resulted in the detection of six birds with ticks attached (Table 4.12). One bird, an Eastern Towhee, was found to have both ticks and bloodborne parasites. I identified all attached ticks as Gulf Coast Ticks (*Amblyomma maculatum*). All ticks were found around the eyes, auricular region, or at the base of the lores. All ticks were also only found during the 2020–2021 winter session. Given haemosporidian parasites are transmitted via dipteran vectors, there is no known relationship between tick haemosporidia prevalence. Feather mite load on all individuals was low (less than one mite per flight feather).

Discussion

My study showed that avian richness and winter capture rates differed between forest stands of different stages. Richness was greatest in the Young Open Canopy stage and decreased with forest age. Previous research has demonstrated that bird richness and abundance during the winter months is highly dependent on forest structure and food availability, as well as snag retention in younger stands and with the age and structural heterogeneity of the surrounding landscape (Childers et al. 1986, Manuwal and Huff 1987). I found that capture rates of wintering passerines varied significantly among different pine stand stages, with the highest rates consistently found in the Young Open Canopy stand. This was not unexpected, as a previous study highlighted greater local abundances in stands within 5 years of initiation (Childers et al.

1986). Conversely, this finding directly contradicts winter research from other forested systems, specifically old-growth conifer forests, where species richness was greatest in old growth forests when compared to younger stands (Manuwal and Huff 1987).

An increased variety of forest ages and conditions surrounding each young stand may support the tendency of winter birds to feed in a variety of locations to meet their caloric needs, with less restricted ranges when compared to breeding individuals who may be confined to a specific territory (Anderson 1980, Childers et al. 1986). Generally, fewer vegetation changes occur between winter and summer months in young stands when compared with mature stands, possibly explaining why young stands can support just as many individuals during the winter as during the summer (Noble and Hamilton 1976). Remaining snags from the previous harvest cycle and residual tree retention as snags may aid in increasing local abundance in these young stands (Iglay et al. 2012, Hanberry et al. 2013).

Within my study, I captured several woodland and canopy feeding birds in the Young Open Canopy stand, including Hermit Thrush and Ruby-crowned Kinglet. Given the affinity for these species to use vertically structured vegetation layers (Dellinger et al. 2020, Swanson et al. 2021), this result was unexpected. Recently disturbed areas and regenerating pine clearcuts have been documented to contain high quality food resources (Greenberg et al. 2011), though supporting data is lacking for the winter months. Early successional and scrub species, such as the three captured sparrow species and House Wren (*Troglodytes aedon*), were captured and observed strictly within the Young Open Canopy stand. While not unexpected, this highlights the importance of maintaining varying age classes and stand conditions within the larger landscape, as species with strict habitat constraints may be limited by the availability of specific conditions when compared to habitat generalists, even during times of limited resources (Diaz et al. 1998).

As a further example, most recaptured individuals in my study were recaptured in the same stand as their original capture. This likely indicates the possibility of high winter site fidelity dictated by a necessity of certain resources and vegetation requirements, even within a large landscape that likely contains multiple stands of similar conditions. The proximity of stands containing varying vegetation structures and food sources likely increased the probability of local site fidelity across time; as stand conditions changed in response to management, disturbance, and weather, individuals were able to continue to find adequate resources within a relatively small area.

Winter site fidelity among resident and migratory passerines is thought to differ among species and ages. Song Sparrows display high fidelity rates with territories year-round (Smith et al. 1996), Ruby-crowned Kinglets display a mix of territoriality and mixed and single species flocking (Somershoe et al. 2009), and some young birds change their overwintering location after their first year (Ketterson and Nolan 1982, Sandercock and Jaramillo 2002). Site fidelity is thought to provide advantages such as increased familiarity with local resources, territorial dominance, and predator avoidance during the winter (Latta and Faaborg 2001, Somershoe et al. 2009, Monroy-Ojeda et al. 2013). In my study, recaptures within season were not unusual, indicating winter site persistence, but only six individuals were recaptured the following winter. With prior studies emphasizing that high intensity and a long temporal period (Sandercock and Jaramillo 2002), or color banding of birds for resighting (Blackburn and Cresswell 2016) were needed to evaluate winter site fidelity for several species, subsequent seasons and studies within this system could prove useful in continuing to evaluate fidelity for many species.

Given the effect of average closing temperature between months and years, temperature is a plausible driver of differences in capture rates between months, even within the same winter season. Thermal heterogeneity has been shown to greatly influence species richness, with areas of high heterogeneity being considered priority areas for conserving breeding and year-round residents (Elsen et al. 2021). Many aspects of forest use and management across the U.S., including wildlife diversity and forest function and productivity, can be influenced by changes in the weather and climate (Joyce et al. 2001). This creates a challenge for researchers and managers alike, with increased focus and preparedness needed for anticipating rapid ecological changes (Glick 2011, Osland et al. 2013).

Isotope data revealed that individuals captured in the Young Open Canopy stand likely captured a larger variety of prey, as illustrated by the greater variation in carbon isotopic values (Figure 4.3). Higher (less negative) carbon isotope values are indicative of dietary assimilation of food sources that were more aligned with C4 producers, such as switchgrass, with lower values (more negative) correlating to C3 producers including pines and shrubs. Woodland species, such as Hermit Thrush, were much more likely to have lower carbon isotope values than species that use a variety of vegetation conditions, such as Ruby-crowned Kinglet. Sparrow species, including Lincoln's, Swamp, and Song, had the highest $\delta^{13}C$ values, indicating their dietary choices are likely different than the remaining captured species. There was a significant difference in $\delta^{13}C$ and $\delta^{15}N$ assimilation in individuals captured in the late stage stand that likely indicates those species are using fewer resources than species captured in the other stands, potentially driven by the smaller amounts of vegetation diversity in the Late Stage stand. However, it is important to recognize that my isotope sample size for that specific stand is very small (n = 2), leading to hesitation in making broad assumptions.

In my study, I was able to show how stable isotope values can be used to provide important wintering data of both resident and migratory species that can be combined with other information to facilitate our understanding of over-wintering avian communities. Using traditional dietary studies, where excrement (guano) reflects ingestion of specific prey (Ralph et al. 1985, Van Horne and Bader 1990, Rytkönen et al. 2018), but not assimilation of nutrients shows how both methods can be used to assess food preference throughout the year. Even though challenges still exist in the interpretation and expanded uses of this isotope data, the values still contribute to our understanding of the winter ecology and basic needs of a wide range of species.

Lower winter parasite prevalence among migratory individuals has been recorded when compared to resident individuals (Soares et al. 2020). From 2,681 blood samples of 51 different avian species, only 2.8 % of all over-wintering migratory individuals were infected with haemosporidian parasites (Soares et al. 2020). It has been suggested that seasonality of vector populations as well as host preference could explain some of this variation between migratory and resident populations. While my sample size was low, my results contradict this prior research; I found a higher prevalence of both bloodborne and external parasites in migratory individuals. My findings also do not align with previous research highlighting the seasonality or temporal decrease of parasitic prevalence through the winter months (Cosgrove et al. 2008, Dunn et al. 2014). With parasite detections in all months except for March, there was no clear pattern of detected prevalence in response to time since breeding season or temperature. However, all documented ticks were found during the 2020–2021 winter season, at a time temperatures fluctuated considerably during the season and monthly rainfall was less on average than in the previous season. Consistently low temperatures are known to decrease the survival of tick nymphs (Dautel et al. 2016) and submersion in water frequently causes death in adult ticks, with species persistence often dependent on survival of the larval stages (Sutherst 1971). Mild winters may lead to increased occurrences of ticks in many environments (Dautel et al. 2016) or increased survival of avian hosts and should be investigated further.

Blood analysis in my study was only conducted through blood smears and a traditional microscopy procedure, which possibly limited my ability to detect parasites due to sampling and observer biases and likely low parasitic prevalence in collected blood samples (Richard et al. 2002, Bell et al. 2015). Soares et al. (2020) used PCR (polymerase chain reaction) based analysis to amplify and target parasite specific mitochondrial DNA, likely leading to higher rates of parasite detections than commonly observed in microscopy-based studies. Future research should focus on collecting additional samples for PCR analysis, which is exceptionally accurate and efficient at detecting bloodborne parasites in recent years due to its ability to detect parasites at low prevalence and the simultaneous detection of multiple species (Freed and Cann 2006, Bell et al. 2015).

Conclusion and Management Implications

Understanding the suite of environmental conditions affecting individuals during the winter months, including local abundance, weather, dietary choices, and parasitic prevalence, are important for any conservation effort focusing on managing forest tracts for migratory and resident bird species. My study highlights the utility of varying stand age and structure within a working pine system as most individuals are likely not dependent on one food source or one forest stage during the winter months. Multiple individuals were found to return to the same over wintering area, illustrating the importance of maintaining vegetation conditions across time, even within vast expanses of similar habitat. More importantly, for many priority resident bird species such as Eastern Towhee (*Pipilo erythrophthalmus*), the Young Open Canopy stage created by recently harvested stands may contain the early successional conditions required throughout the annual cycle. This notion has frequently been documented during breeding studies, but extensive research during the winter months and on both resident and migratory species is missing (Miller and Miller 2004, Miller et al. 2009, Hanberry et al. 2013).

While some data are deficient, particularly how management affects over wintering communities, bird conservation efforts in privately managed pine forests are being recognized for the role the system plays within the larger conservation arena, particularly as other forest systems are lost (Wigley et al. 2000, Hartley 2002, Moorman et al. 2002, Frederickson 2007, Legrand et al. 2007, Miller et al. 2009, Lane et al. 2011, Greene et al. 2016, Greene et al. 2019, Evans et al. 2021). As such, this forested landscape has the potential to support high winter abundances for many avian species. With species responding differently to vegetation characteristics and forest structure differently depending on availability and season, management recommendations for conservation of winter bird diversity may be difficult to make. However, given the propensity of woodland and early successional species to use both open pine-grassland forests following stand initiation and mid to late rotation, thinned stands in this system, efforts should focus on increasing variation in forest structural complexity within mid rotation stands, providing understory structure in late stage, closed canopy stands, and increasing habitat diversity across the managed landscape by maintaining a portion of the area for species that require habitat attributes associated with mature forests. Future research should prioritize sampling vegetation structure and food availability in different forest conditions, movements of individuals throughout the winter months, and investigating the implications of over winter parasitic infections on avian physiology, behavior, and survival.

Tables and Figures

Table 4.1 The models for winter capture rate, degrees of freedom (*df*), log likelihood (logLik), AICc, ΔAIC, and model weight (W_i) for all banding sessions November–March 2019–2022, within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi.

Model	Formula	*df*	logLik	AICc	ΔAICc	W_i
1	AvgCTemp + Location	5	50.934	-89.8	0.00	0.288
2	AvgCTemp + Location + Session	7	53.868	-89.6	0.21	0.259
3	AvgCTemp	3	47.848	-88.9	0.88	0.186
4	Location	4	48.376	-87.4	2.38	0.088
5	Null	2	45.646	-86.9	2.88	0.068
6	Session	4	47.891	-86.4	3.35	0.054
7	AvgCTemp + AvgOTemp	4	47.850	-86.4	3.43	0.052
8	Month	6	47.148	-79.3	10.50	0.002
9	AvgCTemp + Month	7	48.611	-79.1	10.72	0.001
10	AvgCTemp + Location + Month	9	52.068	-78.9	10.86	0.001
11	AvgCTemp + Location + Month + Session	11	55.188	-76.9	12.90	0.000

Table 4.2 The models for winter isotopes, degrees of freedom (*df*), log likelihood (logLik), AICc, ΔAIC, model weight (W_i), and cumulative model weight (Cum. W_i) for all banding sessions November–March 2019–2022, within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi.

Isotope	Model	Formula	*df*	logLik	AICc	ΔAICc	W_i	Cum. W_i
$\delta^{13}C$	1	SpHabitat	4	-59.815	129.5	0.00	0.98	0.98
	2	Species	12	-44.663	137.3	7.79	0.02	1.00
	3	Null	2	-70.278	145.1	15.54	0.00	1.00
	4	Capture Location	4	-68.352	146.6	17.07	0.00	1.00
	5	Capture Location + SpHabitat + Species	14	-44.276	154.7	25.20	0.00	1.00
$\delta^{15}N$	1	Species	12	-14.997	78.0	0.00	0.47	0.47
	2	Null	2	-36.885	78.3	0.30	0.40	0.87
	3	Capture Location	4	-35.625	81.2	3.16	0.10	0.97
	4	SpHabitat	4	-36.628	83.2	5.17	0.03	1.00
	5	Capture Location + SpHabitat + Species	14	-13.611	93.4	15.41	0.00	1.00

Table 4.3 Number of individual birds, by species and forest stage, captured during all banding sessions November–March 2019–2022, within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi.

Species		Alpha Code	Young Open Canopy	Mid Unthinned	Late Stage
American Goldfinch	*Spinus tristis*	AMGO	0	1	0
American Robin	*Turdus migratorius*	AMRO	0	3	0
Carolina Chickadee	*Poecile carolinensis*	CACH	0	0	2
Carolina Wren	*Thryothorus ludovicianus*	CARW	3	6	2
Eastern Phoebe	*Sayornis phoebe*	EAPH	0	1	4
Eastern Towhee	*Pipilo erythrophthalmus*	EATO	3	0	1
Golden-crowned Kinglet	*Regulus satrapa*	GCKI	3	4	1
Hermit Thrush	*Catharus guttatus*	HETH	3	8	8
House Wren	*Troglodytes aedon*	HOWR	2	0	0
Lincoln's Sparrow	*Melospiza lincolnii*	LISP	1	0	0
Northern Cardinal	*Cardinalis cardinalis*	NOCA	1	4	1
Orange-crowned Warbler	*Leiothlypis celata*	OCWA	3	2	0
Ruby-crowned Kinglet	*Regulus calendula*	RCKI	29	20	6
Song Sparrow	*Melospiza melodia*	SOSP	6	0	0
Swamp Sparrow	*Melospiza georgiana*	SWSP	3	0	0
Total Number of Individuals			57	49	25
Number of species			11	9	8

Table 4.4 List of incidental bird species observed by forest stand age, with an "X" denoting detection (auditory or visual), from November–March 2019–2022 within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi.

Species		Young open canopy	Mid Unthinned	Late Stage
American Crow	*Corvus brachyrhynchos*	X	X	
American Robin	*Turdus migratorius*		X	X
Blue Jay	*Cyanocitta cristata*		X	X
Blue-gray Gnatcatcher	*Polioptila caerulea*		X	
Carolina Chickadee	*Poecile carolinensis*	X	X	X
Carolina Wren	*Thryothorus ludovicianus*	X	X	X
Cedar Waxwing	*Bombycilla cedrorum*			X
Chipping Sparrow	*Spizella passerine*	X	X	
Cooper's Hawk	*Accipiter cooperii*	X	X	
Dark-eyed Junco	*Junco hyemalis*	X	X	X
Downy Woodpecker	*Dryobates pubescens*			X
Eastern Bluebird	*Sialia sialis*	X		
Eastern Phoebe	*Sayornis phoebe*	X	X	X
Eastern Towhee	*Pipilo erythrophthalmus*	X	X	X
Fish Crow	*Corvus ossifragus*		X	
Hermit Thrush	*Catharus guttatus*	X		X
Northern Cardinal	*Cardinalis cardinalis*	X	X	X
Northern Harrier	*Circus hudsonius*	X	X	
Orange-crowned Warbler	*Leiothlypis celata*			X
Pileated Woodpecker	*Dryocopus pileatus*		X	X
Pine Warbler	*Setophaga pinus*		X	X
Red-shouldered Hawk	*Buteo lineatus*		X	
Red-tailed Hawk	*Buteo jamaicensis*	X	X	
Red-winged Blackbird	*Agelaius phoeniceus*		X	
Ruby-crowned Kinglet	*Regulus calendula*	X	X	X
Swamp Sparrow	*Melospiza georgiana*	X		X
Tufted Titmouse	*Baeolophus bicolor*		X	X
White-eyed Vireo	*Vireo griseus*		X	
Yellow-throated Vireo	*Vireo flavifrons*			X
Total Number of Species		15	21	18

Table 4.5 Number of species and individuals captured per month, November–March 2019–2022 within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi.

Species		**November**	**December**	**January**	**February**	**March**
American Goldfinch	*Spinus tristis*				1	
American Robin	*Turdus migratorius*		1		2	
Carolina Chickadee	*Poecile carolinensis*	1				1
Carolina Wren	*Thryothorus ludovicianus*	4	3	4		
Eastern Phoebe	*Sayornis phoebe*	2	1	1	1	
Eastern Towhee	*Pipilo erythrophthalmus*		1	1		2
Golden-crowned Kinglet	*Regulus satrapa*	4	1	2	1	
Hermit Thrush	*Catharus guttatus*	5	7	5	1	1
House Wren	*Troglodytes aedon*			2		
Lincoln's Sparrow	*Melospiza lincolnii*	1				
Northern Cardinal	*Cardinalis cardinalis*		1	2		3
Orange-crowned Warbler	*Leiothlypis celata*		2	3		
Ruby-crowned Kinglet	*Regulus calendula*	19	11	16	5	4
Song Sparrow	*Melospiza melodia*	3	1	2		
Swamp Sparrow	*Melospiza georgiana*	2	1			
Number of Individuals		41	30	38	11	11
Number of Species		9	11	10	6	5

Table 4.6 Original capture dates and locations, and recapture dates and locations for 19 individual birds, for a total of 21 recaptures November–March 2019–2022 within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi.

Species	Capture Date	Capture Location	Recapture Date	Recapture Location	Recapture Distance (m)	Second Recapture Date	Second Recapture Location	Recapture Distance (m)
Carolina Wren	1/13/2021	Mid Unthinned	1/11/2022	Mid Unthinned	116.30			
Carolina Wren	11/10/2021	Mid Unthinned	1/11/2022	Mid Unthinned	125.40			
Eastern Phoebe	11/5/2020	Late Stage	2/3/2021	Late Stage	109.99			
Golden-crowned Kinglet	11/4/2020	Young Open Canopy	1/13/2021	Young Open Canopy	79.62			
Hermit Thrush	11/13/2019	Late Stage	12/2/2020	Late Stage	38.00			
Hermit Thrush	11/13/2019	Young Open Canopy	1/12/2021	Young Open Canopy	177.48	1/12/2022	Young Open Canopy	177.48
Hermit Thrush	11/3/2020	Late Stage	12/1/2020	Mid Unthinned	199.94			
Hermit Thrush	11/4/2020	Mid Unthinned	2/3/2021	Mid Unthinned	57.82			
Hermit Thrush	12/7/2021	Late Stage	12/8/2021	Late Stage	85.92			
Ruby-crowned Kinglet	11/13/2019	Mid Unthinned	12/12/2019	Mid Unthinned	0.00			
Ruby-crowned Kinglet	12/11/2019	Mid Unthinned	12/12/2019	Mid Unthinned	125.40			
Ruby-crowned Kinglet	1/15/2020	Mid Unthinned	2/4/2021	Mid Unthinned	0.00			
Ruby-crowned Kinglet	1/12/2021	Young Open Canopy	1/14/2021	Mid Unthinned	289.93	2/4/2021	Young Open Canopy	138.22
Ruby-crowned Kinglet	1/12/2021	Mid Unthinned	1/14/2021	Mid Unthinned	57.82			
Ruby-crowned Kinglet	1/12/2021	Young Open Canopy	1/14/2021	Young Open Canopy	0.00			
Ruby-crowned Kinglet	1/14/2021	Young Open Canopy	2/3/2021	Late Stage	74.94			

Table 4.6 (continued)

Species	Capture Date	Capture Location	Recapture Date	Recapture Location	Recapture Distance (m)	Second Recapture Date	Second Recapture Location	Recapture Distance (m)
Ruby-crowned Kinglet	1/14/2021	Young Open Canopy	12/8/2021	Young Open Canopy	0.00			
Ruby-crowned Kinglet	11/10/2021	Mid Unthinned	3/1/2022	Mid Unthinned	57.82			
Song Sparrow	12/2/2020	Young Open Canopy	1/13/2021	Young Open Canopy	79.62			

Table 4.7 Capture rates for each bird banding session, by forest stand stage and month, November–March 2019–2022 within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi. Rate is reported as number of captured individuals per net hour.

Year	Month	Young Open Canopy	Mid Unthinned	Late Stage	Total
2019	November	NA	0.05	0.17	0.11
	December	0.29	0.11	0.03	0.10
2020	January	NA	0.03	0.02	0.03
	February	NA	NA	NA	NA
	March	NA	0.01	0.10	0.02
	November	0.18	0.08	0.03	0.21
	December	0.08	0.16	0.02	0.11
2021	January	0.19	0.07	0.00	0.27
	February	0.01	0.04	0.04	0.11
	March	0.00	0.09	0.02	0.03
	November	0.03	0.00	0.00	0.08
	December	0.10	0.03	0.05	0.09
2022	January	0.05	0.01	0.00	0.07
	February	NA	NA	NA	NA
	March	0.07	0.01	0.00	0.06
	Total	0.08	0.06	0.03	0.05

Table 4.8 Fitted Linear Model (LM) results for predicting capture rate in winter months from November–March 2019–2022, within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi. Model predictor variables were location (Young Open Canopy, Mid Unthinned, or Late Stage), average closing temperature (AvgCTemp, in °C), and Session (1 [2019–2020], 2 [2020–2021], or 3 [2021–2022]). Response variable for all models in (a) was capture rate (number of individuals captured per net hour [sum of number of hours each net was open, grouped by stand stage]). Bolded P-values indicate statistical significance (α = 0.05).

		Estimate	Std. error	t value	P-value	□AICc	Cumulative Model Weight	R^2
Model 1	Intercept/Late	0.09	0.03	3.06	**0.00**	0	0.46	0.26
	Mid	0.02	0.02	0.82	0.42			
	Young	0.08	0.03	2.42	**0.02**			
	AvgCTemp	-0.00	0.00	-2.21	**0.03**			
Model 2	Intercept	0.04	0.02	2.08	**0.05**	0.88	0.75	0.14
	Mid	0.02	0.03	0.79	0.45			
	Young	0.06	0.03	2.31	**0.03**			

Table 4.9 Fitted Linear Model (LM) results for predicting isotopic δ^{13}C and δ^{15}N in 26 avian blood samples from birds captured during the winter months from November 2019–December 2020 within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi. Predictor variables included species habitat association and species (in alpha code). Bolded P-values indicate statistical significance ($\alpha = 0.05$).

Isotope	Model		Estimate	Std. error	95% CI	t value	*P* value
δ^{13}C	1	Intercept	-25.930	2.568	-30.962, -20.900	-10.099	< **0.001**
		Early Successional	6.006	2.706	0.701, 11.310	2.219	**0.037**
		Woodland	0.386	2.647	-4.801, 5.573	0.146	0.885
δ^{15}N	1	Intercept	4.430	0.567	3.319, 5.542	7.811	< **0.001**
		CARW	-0.090	0.802	-1.662, 1.482	-0.112	0.912
		EAPH	1.115	0.695	-0.246, 2.476	1.605	0.129
		EATO	-0.420	0.802	-1.992, 1.152	-0.524	0.608
		GCKI	-1.530	0.802	-3.102, 0.042	-1.908	0.076
		HETH	-0.053	0.606	-1.241, 1.135	-0.087	0.932
		LISP	0.560	0.802	-1.012, 2.132	0.698	0.496
		NOCA	-2.790	0.802	-4.362, -1.218	-3.478	**0.003**
		RCKI	0.338	0.613	-0.862, 1.539	0.552	0.589
		SOSP	1.095	0.695	-0.266, 2.456	1.576	0.136
		SWSP	1.463	0.655	0.180, 2.747	2.234	0.041
	2	Intercept	4.669	0.200	4.277, 5.060	23.350	< **0.001**

Table 4.10 δ13C and $\delta^{15}N$ for 26 birds captured during the winter months November 2019–December 2020, from within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi. Species means (or single value where applicable) and standard deviations (NA when only one sample) in parentheses are included as well as species alpha code (Pyle and DeSante 2003), number of samples per species, and ecosystem association.

Species	Alpha Code	N	Habitat	$\delta^{13}C$	$\delta^{15}N$
American Robin	AMRO	1	Urban	-25.93 (NA)	4.43 (NA)
Carolina Wren	CARW	1	Early Successional/ scrub	-25.15 (NA)	4.34 (NA)
Eastern Phoebe	EAPH	2	Woodland	-24.57 (0.45)	5.55 (0.26)
Eastern Towhee	EATO	1	Early Successional/ scrub	-25.47 (NA)	4.01 (NA)
Golden-crowned Kinglet	GCKI	1	Woodland	-25.67 (NA)	2.90 (NA)
Hermit Thrush	HETH	7	Woodland	-25.12 (0.87)	4.38 (0.33)
Lincoln's Sparrow	LISP	1	Early Successional/ scrub	-17.27 (NA)	4.99 (NA)
Northern Cardinal	NOCA	1	Early Successional/ scrub	-21.70 (NA)	1.64 (NA)
Ruby-crowned Kinglet	RCKI	6	Woodland	-26.34 (1.45)	4.77 (0.74)
Song Sparrow	SOSP	2	Early Successional/ scrub	-16.26 (0.59)	5.53 (0.12)
Swamp Sparrow	SWSP	3	Early Successional/ scrub	-19.07 (3.98)	5.89 (0.82)

Table 4.11 Confirmed parasitic infections in individuals captured November 2019–March 2022 from within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi. Capture locations were Young (Young Open Canopy), Mid (Mid Unthinned), or Late (Late Stage); migratory status was either resident or migrant.

Species	Migratory Status	Collection Date	Capture Location	Haemoproteus	Trypanosoma
Eastern Towhee	Resident	1/13/2021	Young	X	
Ruby-crowned Kinglet	Migrant	2/4/2021	Late	X	
Golden-crowned Kinglet	Migrant	11/10/2021	Mid		X
Ruby-crowned Kinglet	Migrant	12/7/2021	Young	X	

Table 4.12 Positive tick survey results from birds captured during the winter months from November 2019–March 2022 from managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi. Migratory status was resident or migrant.

Avian Species	Migratory Status	Number Surveyed	Tick Prevalence	Tick Species
Hermit Thrush	Migrant	19	5%	Gulf Coast Tick
Lincoln’s Sparrow	Migrant	1	100%	Gulf Coast Tick
Song Sparrow	Migrant	6	17%	Gulf Coast Tick
Eastern Towhee	Resident	4	25%	Gulf Coast Tick
House Wren	Migrant	2	50%	Gulf Coast Tick
Carolina Wren	Resident	11	9%	Gulf Coast Tick

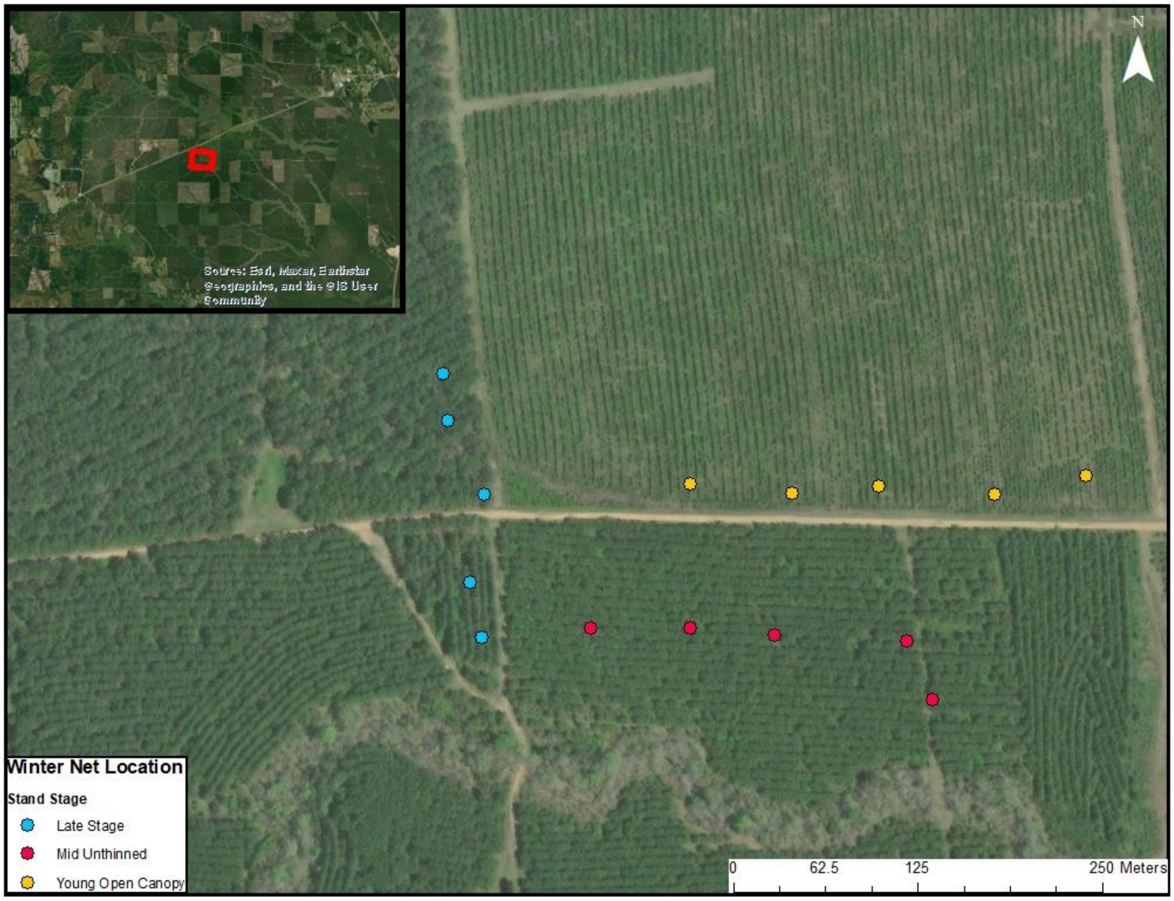

Figure 4.1 Map of winter mist net placement within loblolly pine (*Pinus taeda*) stands in east-central Mississippi. Inset map shows location of chosen managed loblolly pine stands within the study site in east-central Mississippi.

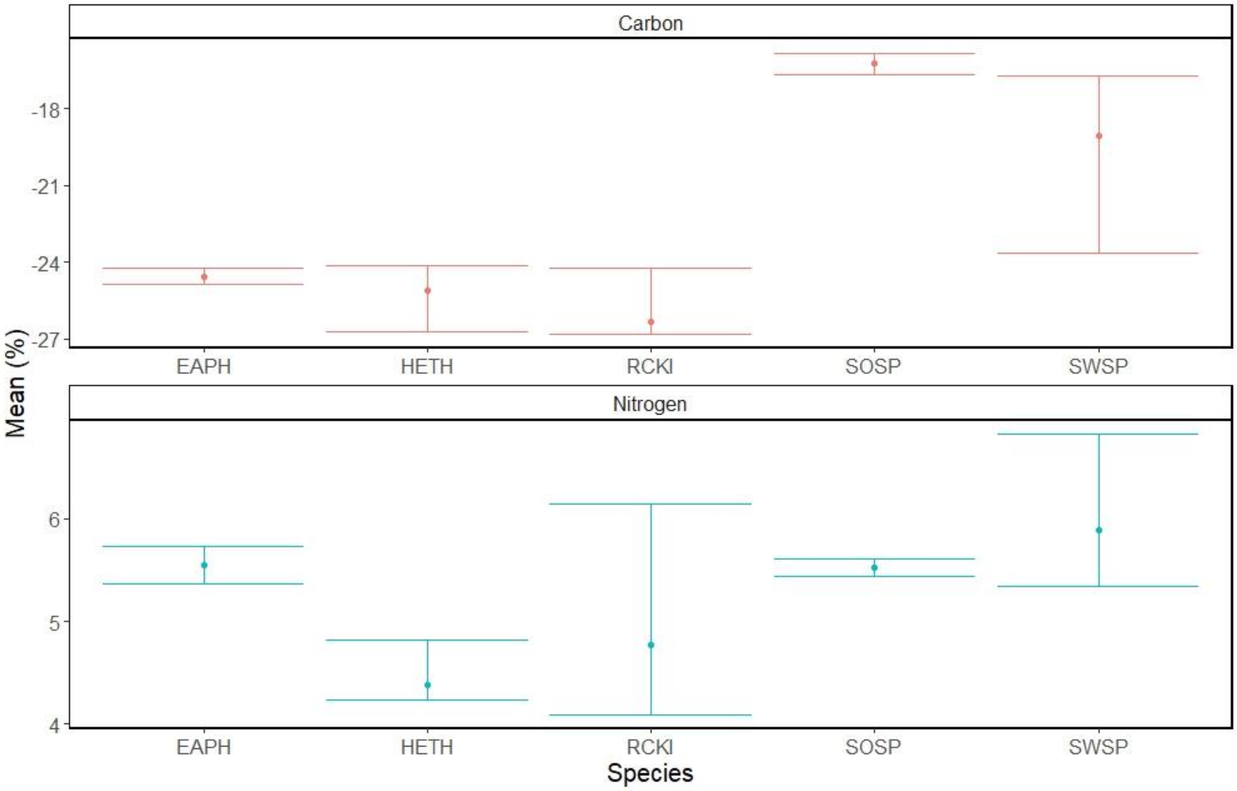

Figure 4.2 Mean $\delta^{13}C$ (Carbon) and $\delta^{15}N$ (Nitrogen) values, with maximum and minimum values, for species with 2 or more captures, captured during the winter months from November 2019–December 2020 within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi.

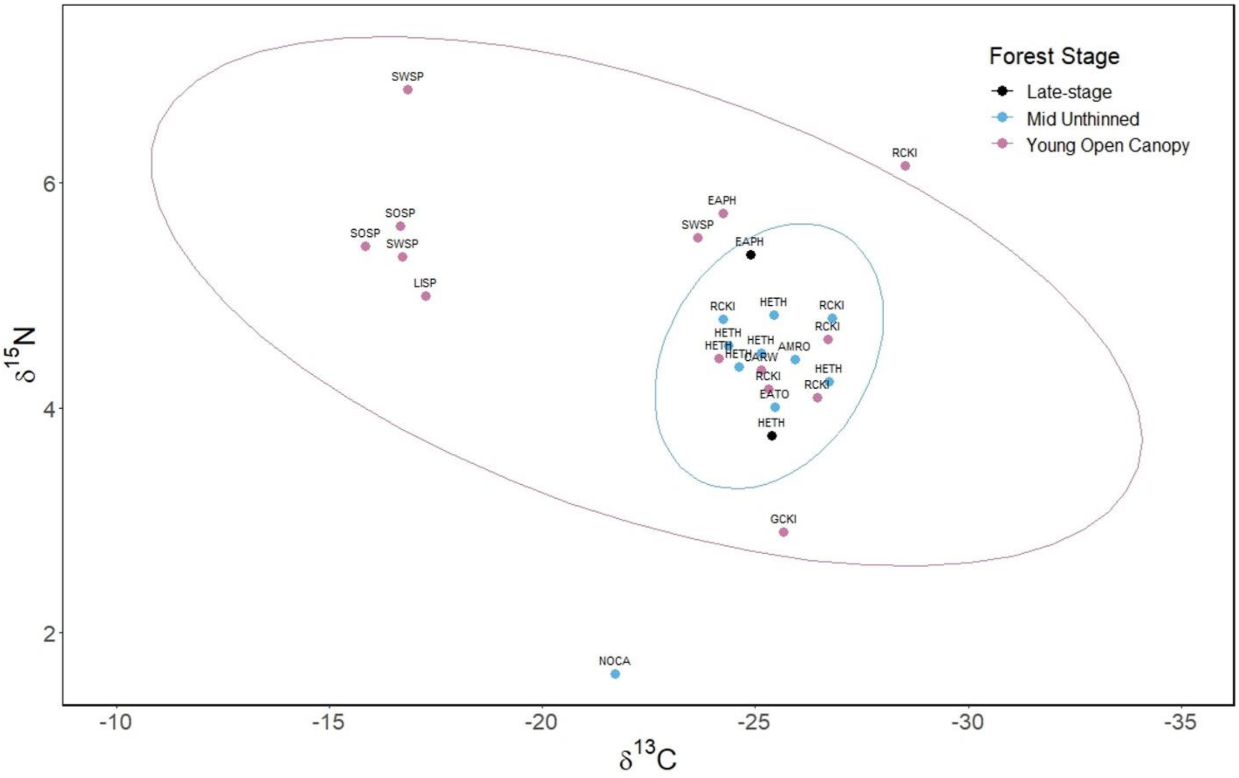

Figure 4.3 Blood isotope values for δ^{13}C and δ^{15}N from 26 birds captured during the winter months from November 2019–December 2020 within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi. Ellipses represent capture location, with a 95 % confidence interval. Each labeled point represents a different captured bird, with multiple captured individuals of some species. Refer to table 4.3 for species abbreviations.

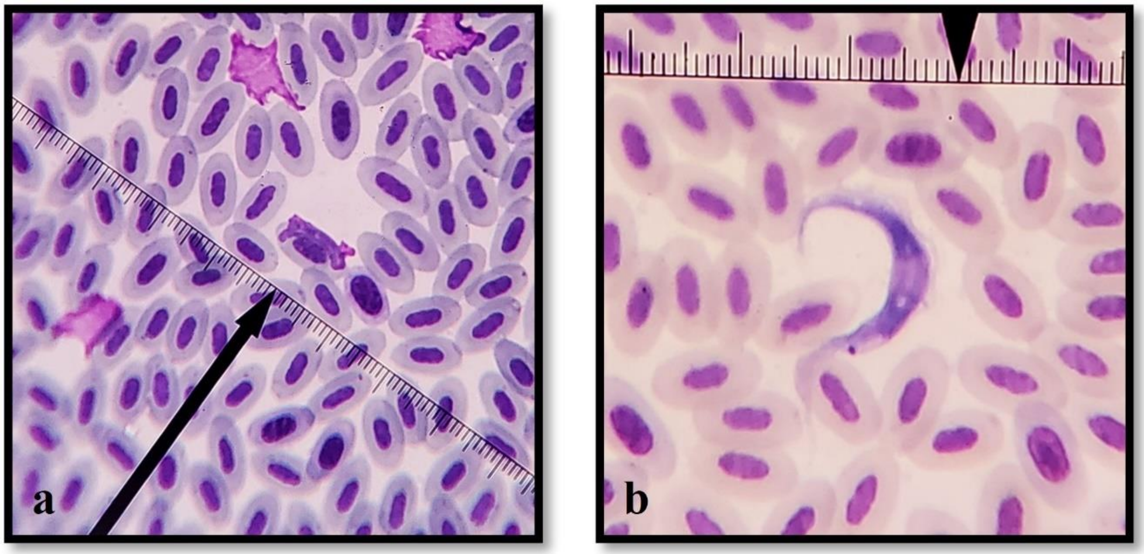

Figure 4.4 (a) Haemoproteus spp. observed in a Ruby-crowned Kinglet captured in February 2021, and (b) Trypanosoma spp. observed in a Golden-crowned Kinglet captured in November 2021. Both birds were captured within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi. Images were taken under 1000x magnification and oil immersion.

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CHAPTER V

USE OF A PRIVATE, WORKING FOREST BY CHUCK-WILL'S-WIDOW, A SPECIES FOR WHICH LITTLE CONTEXT FOR EFFECTIVE CONSERVATION CURRENTLY EXISTS

Forest management practices often aim to increase early successional conditions for breeding birds by attempting to mimic natural disturbances such as fire, and can benefit aerial insectivores (Tozer et al. 2014, Farrell et al. 2017). Chuck-will's-widow (*Antrostomus carolinensis*; hereinafter, CWWI), one such species, is commonly found breeding throughout the southeastern United States, often within pine ecosystems. Little is known about CWWI's life history, including survival rates, population densities, physical aspects of location used for nesting and foraging, and how CWWI use and are affected by managed landscapes (Straight and Cooper 2012). Like most nightjar (family Caprimulgidae) species, only approximate ranges are known, and little information exists on migratory connectivity (Holyoak 2001, Ng 2018).

CWWI are a crepuscular species, typically active only in the early morning hours and around sunset (Mills 1986, Evens et al. 2020, Straight and Cooper 2020). They are known to forage low to the ground, with most of their known diet consisting of moths and beetles (Bent 1940). However, CWWI are also known to opportunistically take small birds such as warblers, sparrows, and hummingbirds as prey, as well as amphibians (Bent 1940, Rohwer and Butler 1977, Straight and Cooper 2020). Like other nightjars, CWWI activities are thought to be influenced by the amount of moonlight and lunar phase, with a longer activity period, and more songs and forages, noted during full moons and increased moonlight intensity (Leopold and Eynon 1961, Mills 1986, Vilella 1995, Reino et al. 2015, Evens et al. 2020).

Previous studies have assessed nest site selection by CWWI, but conflicting reports exist, and nest site selection may vary by geographic location and availability (Ganier 1964, Imhof 1976, James and Neal 1986, Peterjohn and Rice 1991, Palmer-Ball 1996, O'Connor 2013). Recent studies indicated that CWWI nests were found in areas of less canopy cover. Considering CWWI are visual foragers, it was hypothesized that this occurrence could be due to the increased light levels in open and early successional forest conditions (Cooper 1981, O'Connor 2013). Similarly, nest sites were found to be closer to forest edges, often under dense cover, near old roads (Imhof 1976, O'Connor 2013).

Within their known range, the distribution and abundance of CWWI is likely dependent on resources not contained within a single forest stand or patch type, with local distributions and territories being influenced by characteristics of available patches and characteristics of the surrounding patches, resources such as food supply, and competitive pressures (Sisk et al. 1997, Ries and Sisk 2004, Wilson and Watts 2008). Regenerating forest stands that create edges within the larger forested landscapes, a management practice that is common within the CWWI breeding range, was shown to have a positive effect on nightjars, possibly through greater access to prey (Summerville and Crist 2002, Wilson and Watts 2008). In a similar manner, newly planted forest stands adjacent to mature stands in intensively managed forest landscapes may provide CWWI and similar species foraging and nesting habitats in close proximity.

Forest management practices that aim to increase early successional conditions also appear to benefit Eastern Whip-poor-wills (*Antrostomus vociferus*, hereinafter EWPW), a similar species (Tozer et al. 2014, Farrell et al. 2017). Whereas there is much overlap in resource use during the breeding season between CWWI and EWPW, there appears to be an expansion in range for EWPW into areas where CWWI are more prevalent but declining, and an expansion of CWWI north into areas typically dominated by EWPW (Cooper 1982). With much of the landscape considered suitable for CWWI in the southeast on privately owned forestlands, forested landscapes altered by more active regeneration forest management schemes may provide these species with opportunities not found in less intensively managed forests (Wilson and Watts 2008).

In these managed pine forests, which are characterized by dense ground cover in regenerated stands and temporally open midstories in thinned, mid to late stage (10–15 years prior to harvest) stands, CWWI are among a small group of endemic avian species (Jackson 1988, Platt et al. 1991). Even though a cumulative population decline of approximately 69% for CWWI since 1966 has been noted, no significant conservation action plan has been developed for this species (Sauer et al. 2017). Current Partners in Flight rankings do not indicate the species is of high conservation concern, with a score of 2 out of 5 (5 being the highest concern), but a score change has been recommended due to crashing populations and increasing threats to breeding and non-breeding habitats including increasing urbanization and altering of landscapes (Panjabi et al. 2020). In general, there are likely many factors that can be attributed to this decline, including increased use of pesticides, land-use changes, and increased amounts of urbanization within their historical range (Straight and Cooper 2020). CWWI is listed as near threatened by the International Union for Conservation of Nature (Birdlife International 2020) and is listed as a priority species by the East Gulf Coastal Plain Joint Venture (Greene et al. 2021).

At the time of this research, there were no known studies focusing on CWWI use within working pine forests in the southeastern U.S. during the breeding season. Given the potential for working forests to provide resources supporting this species, research was needed to address questions regarding occupancy and abundance in relation to the amount and type of forest within the larger managed forest. Therefore, I examined how the amount (meters squared) and age of managed loblolly pine (*Pinus taeda*) forest at different spatial scales surrounding designated survey points would influence CWWI abundance, and if lunar illumination and cloud cover would influence detectability. Younger open stands are likely providing foraging opportunities, but forest cover nearby could be crucial for nesting sites. I used distance based point count data collected at designated survey points to estimate CWWI densities for two summer breeding seasons. I created a new capture method for CWWI, as there is no specific method proven successful for capturing this unusual species and evaluated captured CWWI parasite prevalence and isotopic nutrient assimilation for captured individuals.

I predicted that an increase in lunar illumination would lead to an increased ability to detect CWWI, regardless of the amount and type of surrounding forest cover. However, I considered recent thinning events could lead to a temporary reduction in detection ability due to an increase in disturbance. I predicted that CWWI detections would occur most frequently in mid rotation, thinned pine stands, but that abundance would increase in response to increasing amounts of early successional conditions within all buffers. This could be anticipated to create connectivity between potential nesting and roosting sites and appropriate open foraging conditions. I expected CWWI densities to be high when compared to studies in other systems (James and Neal 1986, Baumgartner and Baumgartner 1992, Hayes et al. 2010), given my pilot surveys in the summer prior to initiating standardized surveys. I anticipated that traditional passerine mist netting methods, specifically mesh size, might prove ineffective for successful capture of this species, and that modifications to traditional playback based target netting would be needed. Finally, I predicted isotopic blood carbon and nitrogen would show little variation between individuals and would heavily correlate to pine systems. Understanding the relationship between CWWI occupancy and forest management practices is important to the continuing conservation efforts for the declining species.

Methods

Study Site

See Chapter 2 of this dissertation for a full description of study area.

Abundance Surveys

CWWI are known to arrive on the breeding grounds in Florida, Texas, and South Carolina as early as April, with incubation beginning shortly after (Straight and Cooper 2020). As soon as CWWI arrived in Mississippi, typically in late April and early May (as documented by eBird checklists [eBird 2017]), I began standardized surveys and trapping protocols. I completed pilot surveys in May and June of 2019 to determine appropriate timing for future standardized surveys. In 2020 and 2021, I began surveying in mid-May, coinciding with the start of the long-term summer avian point count surveys (Chapter 2). I established survey locations at 750-meter increments along timber roads within Kemper County (Figure 5.1); I used this distance to establish independence between individuals, assuming that CWWI have a similar home range size to EWPW, and that like EWPW, individual CWWI roost in the same general location each day (Bent 1940, Holyoak 2001, Tozer et al. 2014). I gave each survey point a priority ranking, designated by amount of surrounding Weyerhaeuser Company property. I began surveying 30 minutes after sunset and continued for 20 minutes. Given these protocols, each observer could only survey one point each survey night.

Each survey was conducted by one observer, either me or a trained technician, at each location, and with each point surveyed twice during each breeding season. I assumed all detected auditory responses were male CWWI because female CWWI are not known to sing (Straight and Cooper 2012). I also assumed that individuals exhibited site fidelity during the season as male nightjars have been documented to day roost near their mate's chosen nest site (Lack 1932, Weller 1958). I recorded the approximate distance and direction of each detected CWWI, at the time of first detection within the survey period. I recorded survey conditions during each survey. Specifically, I recorded survey start and end time, temperature, cloud over, moon phase, and wind speed. I also retrieved daily rain totals from the nearest weather station (Macon, MS) through the National Weather Service NOWData online database (NOAA 2022). I made note of detections of similar species during the surveys; Common Nighthawks (*Chordeiles minor*, hereinafter CONI) are a common species that has been observed at the same study site while EWPW were detected in low numbers during pilot surveys.

Target Netting

Trapping attempts occurred between May and June from 2019–2021. I first considered target netting sites based on CWWI activity on nightly point counts. The selection of the specific net location then depended upon several characteristics including confirmed CWWI occupancy, surrounding forest stand conditions, proximity to potential foraging sites, and the substrate where the nets and decoy would be placed. My pilot surveys indicated that CWWI exhibit a local preference for thinned stands within the managed pine forest. I passively observed birds sitting on the edge or middle of roads with a layer of gravel on the surface as well as on cleared log-roads comprised of clay and soil. To mimic this inclination, I chose sites with similar characteristics to place my decoy and net.

During pilot efforts in 2019, I used a standard 12-meter, 30 mm mist net with a Foxpro Wildfire 2 (FOXPRO Inc.) speaker. I pre-loaded the speaker with a CWWI song, and volume could be controlled via remote in the field during operation. This allowed me to power the speaker on and off and vary the call from a remote location during the targeted trapping period. Nets were oriented diagonally across clay and gravels roads on or near the edge of thinned mid-age to late rotation pine stands at least 15 years of age (as determined by the early pilot survey results; Figure 5.2) at least 15 minutes prior to sunset.

Trapping attempts during my pilot season (2019) without a decoy led to a reduced willingness by birds to approach the net with audio lure alone. Therefore, I created a fabricated decoy CWWI to entice individuals to approach my nets at a lower point close to the ground instead of remaining in the trees above the net. I constructed the decoy by scanning a Chuck-will's-widow specimen on loan from the Mississippi Museum of Natural Science using the Qlone 3D Scanner smartphone app. I then printed a 3-D copy using a LulzBot TAZ 6 3-D printer. I covered the printed model with a thin layer of air-drying modeling clay and hand-painted it with brown paint (Figure 5.3).

In May 2020, I coupled the mist net with the decoy, and attempted configuring two nets in a 'V' formation with the speaker and decoy in between the legs of the V. In June of 2020, I increased my net mesh size to 60 mm, and used multiple net configurations including a single net and a V with both a 30 mm and 60 mm mist net. I deployed the decoy 0.5–2 m from the net, on the side opposite from chosen occupied pine stand. I placed the speaker 0.5–1 m behind the decoy, typically in a small ditch or grass patch for added camouflage.

As an alternative method, I attempted to trap individuals using the decoy and a bownet. Bownets create a brief dome over a resting bird or nest and are an effective tool for catching diurnal raptors and shorebirds and other ground nesting birds in open areas (Hull and Bloom 2001, Woodin et al. 2012, de Zwaan et al. 2018, Gratto-Trevor 2018). Given the behavior of CWWI to land on the ground and approach the speaker and decoy, I attempted to use a bownet as an alternative option for capture, if an individual came close enough to the decoy, speaker, and net.

Finally, in 2021, I used a standard 60 mm, 12-meter nylon mist net with added latex hose rings attached to the mist net loops on 3.1 m conduit poles (Figure 5.4). I chose latex hose because it is designed to be used as an elastic band and shock absorber, is easily cut into the chosen length, and is durable and would remain flexible at a variety of temperatures. The added trammel rings were comprised of 0.6 cm latex hose, cut into 17.8 cm pieces that I attached to each trammel end-loop. The rings were large enough to go around the mist net poles yet loose enough to stretch when the net moved. I slid the hollow poles onto rebar hammered into the ground to provide stability without sacrificing height. I tied guy lines 2/3 of the way up the poles and staked either parallel or slightly angled from the poles to prevent the net from sagging and to maintain tension. These added latex loops allowed me to open the mist-net during normal operation like an unmodified mist-net and could be easily adjusted in the field as needed. The use of the added loops did not prevent the rapid setup or take down of the mist-net.

I erected the mist net 2–4 m from the corner of the targeted pine stand, on a gravel and clay logging road. I positioned the lower trammel 0.25 m from the ground to prevent individuals from flying or walking under the net. I arranged the decoy on the ground 0.5–2 m from the net, on the side opposite from the targeted pine stand. I placed the speaker 0.5–1 m behind the decoy, typically in a small ditch or grass patch for added camouflage.

Caprimulgids are visual foragers (Woods and Brigham 2008). As such, I considered that attracting CWWIs to my target trapping site too early (relative to sunset) would allow birds to better see and potentially avoid nets. I commenced playback at least 20 min after sunset in locations where I had a singing individual within a 300 m radius. Even with cloud cover, this provided enough light for me to observe their behavior while still reducing the birds' ability to see the nets. I used playback until the targeted individual was caught or up to 1 hr after playback initiation. Overall, I kept most target-netting sessions brief (5–45 min), dependent upon continued bird response. If an individual responded but was not captured, I returned to the location at least one week after the first attempt. I documented all behavioral observations (e.g., tail fanning, bill clacking, head bobbing, and grunting; Holyoak 2001) when individuals were able to be observed during trapping events.

Banding and Blood Samples

I processed all individuals within 10 minutes of capture, and after closing the net, released them near the point of capture. After confirmed capture, I quickly removed the individual from the mist net, and turned off the playback. I took a GPS location at each capture location, noted capture method, time to capture, and recorded environmental conditions including cloud cover, wind speed, temperature, and moon phase. Upon retrieval of the bird, I followed standard banding procedures (USGS permit #23835-F). I fitted each captured bird with a USGS uniquely numbered leg band, aged, sexed, massed, measured for wing chord (resting, or not flattened) and examined for fat and muscle development. Birds were aged and sexed according to the species account in the Pyle Guide (1997). I also examined each bird for a brood patch or cloacal protuberance. If an individual had a brood patch, it was assumed to be female (Pyle 1997). If an individual was lacking a brood patch and cloacal protuberance, I used plumage structure and feather coloration patterns to identify male versus female.

I took a blood sample from the brachial vein of each bird, using a sterile 27-gauge needle. I collected all blood samples in non-heparinized capillary tubes (volume = 74 µl) and obtained less than 1% body weight recommended as a best practice (Fair et al. 2010). I placed the collected blood in a 1.2 mL cryogenic externally or internally threaded storage vial. Blood samples were immediately placed on ice, and upon return to campus frozen at -80 C. After freeze-drying, blood samples were sent for dietary isotope analysis to The Center for Applied Isotope Studies at the University of Georgia (Athens, Georgia) for isotope analysis of carbon ($^{13/12}$C, hereinafter δ^{13}C) and nitrogen ($^{15/14}$N, hereinafter δ^{15}N). I then compared δ^{15}N and δ^{13}C values from the CWWI to previously collected switchgrass (*Panicum virgatum*), loblolly pine, and rubus (*Rubus argutus*) samples (provided by Marshall 2016 and Fuller-Morris 2018). A previous study conducted at the same location concluded that it was possible for individuals of a forest-dwelling species (Yellow-breasted Chat, *Icteria virens*) to alter their diet based on the availability of some plant species (Fuller-Morris et al. 2020). Considering sampled forest stands differed by rotation stage, identifying correlations between CWWI and vegetation carbon and nitrogen levels could provide further insights of CWWI foraging habitats.

Statistical Analyses

All analysis was completed in ArcMap (Esri ArcGIS Desktop 10.8.1) and R version 4.0.3 (R Core Team 2020). All plots were created using the *ggplot2* package (Wickham 2016). Local landscape data were provided by Weyerhaeuser Company as a geospatially explicit forest stand shapefile. The file included stand age and initiation date, size (in acres), management actions and associated dates of management, predominant managed species, and stand use for all managed stands, unmanaged areas such as logging decks, and streamside management zones (SMZ).

Abundance Models

I determined the location of each detected CWWI by combining the distance and direction values from each survey location with the location of the survey point using the *Geosphere* package in R (Hijmans 2019). I combined the calculated CWWI detection locations with the landscape shapefile in ArcMap. I completed a spatial join to identify occupied stands. I calculated basic statistics including range and mean age of stand where CWWI were detected. I completed proximity analyses for each detection; I determined proximity of survey points to young open canopy pine stands (0–5 years since initiation) and pine stands greater than 15 years since initiation using a geodesic distance calculation and determined distance of detection for forest edge.

To assess scale of effect, I created radial buffers of 375-meters, 750-meters, and 1125-meters around each survey point in ArcGIS. I used the 375-meter buffer to split the distance between survey points and identify the specific set of forest characteristics around each survey point, the 750-meter buffer to include all values around each point without overlapping the neighboring point, and 1125-meter to include the nearest points within a distance of 1.5 times the unique point values, a distance of which is greater than the assumed breeding territory size of the species. Using the spatial join, intersect, and calculate geometry functions in ArcGIS, I identified the amount of area of each individual stand within each buffer both as a percentage of the total buffer area and as an amount in meters squared. I then grouped the calculated stand data by age and by whether or not the stand was thinned. To determine amount and length of forest edge, I created a negative 50 m buffer around each managed pine stand, overlaid the new layer with the original forest stand layer, and erased the overlapping area. I chose 50 m as the forest edge width to follow previous research demonstrating edge effects until that distance into each forest stand on a variety of taxa, though some birds are known to be influenced by forest edge widths deeper than 50 m (e.g., Matlack 1993, McCollin 1998, Young and Mitchell 1994, Winter et al. 2000, Vetter et al. 2013, Bereczki et al. 2015, Terraube et al. 2016); limited research exists regarding edge effects and nightjars (Wilson and Watts 2008, Akresh and King 2016). I joined the negative 50 m buffer layer with each of the three buffer layers by intersecting the layers and joined the edge values by survey point number. I created three forest edge categories: total forest edge, unthinned forest edge, and thinned forest edge.

I used single season N-mixture models using the *pcount* function within package *unmarked* in R to account for heterogeneous detection probabilities and estimate abundance of CWWI for each season (Fiske and Chandler 2011). Model covariates included Julian date, temperature in degree Celsius, wind speed in kilometers per hour, estimated percent cloud cover, and percent lunar illumination as observation, or detection, covariates, and site covariates including percentages and amounts (in meters squared, scaled) of managed pine forest of different age classes and amount (in meters squared) and length (meters) of forest edge within each buffer of each survey point within 375-meter, 750-meter, and 1125-meter buffers of each survey point (Table 5.1). The response variable was number of detections (the number of individuals identified) per survey point per survey night. I identified covariates likely to be significant based on prior research and personal observations, ran single covariate models with all identified covariates, and then modeled all possible combinations of the selected covariates (Table 5.2). I also ran single covariate models with the remaining covariates to identify any unexpected effects. I ranked models by second order Akaike Information Criterion (AICc; AIC corrected for small sample size) and retained models within an AIC of 2 or less of the top model. I ran a MacKenzie and Bailey goodness-of-fit test on each competing model, which calculates the Pearson's chi-square fit statistic from the observed and expected frequencies of detection, in the *AICcmodavg* package, and included 1000 bootstrap samples (Mazerolle 2020). I determined a model was fit appropriately if the chi-square P was ≥ 0.1 (MacKenzie and Bailey 2004). Covariates were considered statistically significant at $\alpha < 0.05$.

I retrieved yearly abundance estimates using *ranef*; by summing all columns, I was able to estimate posterior distributions of the random variables by year, resulting in estimates of abundance (Fiske and Chandler 2011). I used *confint* to extract 95% confidence intervals from included covariates (R Core Team 2020). I created a weighted mean of estimates created by competing models. I then converted the weighted mean to a density estimation using the number of hectares within the survey area.

Results

Breeding Abundance and Density

I detected CWWI at all completed night survey points (Figure 5.5). After two years and surveying each point twice during the breeding season, I had 295 detections of CWWI across 49 survey points. I detected on average more individuals per survey point in 2020 than in 2021 (mean ± SD; 3.3 ± 1.67, 2.9 ± 1.60 respectively). In 2020, I detected 132 individuals in unthinned pine stands, while I detected 11 CWWI in thinned stands at least 15 years since stand initiation. In 2021, I detected 82 individuals in unthinned pine stands while 26 were detected in thinned pine stands 15 years since initiation. Most CWWI were typically detected within 55 m of a forest stand edge. The remaining individuals were determined to be detected while in logging decks, non-pine forested areas (such as SMZ), and non-Weyerhaeuser forested areas.

I also detected CONI and EWPW during surveys in both years. I detected 14 EWPW and five CONI in 2020, and 13 EWPW and four CONI in 2021. Common Nighthawk often began calling prior to survey start time, indicating that our detection values could be underestimated and should not be used to infer local abundance. Given the range of dates of detection of EWPW (May 19–June 30), individuals likely were not detected during spring or fall migration and could be breeding at the site. Other nocturnal or crepuscular species detected in 2020 were Barred Owl (*Strix varia*) and Eastern Screech-Owl (*Megascops asio*), and Eastern Screech-Owl and Great Horn Owl (*Bubo virginianus*) in 2021.

Top models for 2020 (Table 5.3) indicated that as the percent of thinned managed pine 16–20 years of age within a buffered distance of 1125 meters around each survey point increased, the predicted abundance of CWWI would decrease ($\beta = -0.230$, SE = 0.11, $P < 0.05$; Figure 5.6). Conversely, in 2021, top models identified the percent of pine 0–5 years since stand initiation within a buffered distance of 375 meters from each survey point positively influenced CWWI abundance estimates ($\beta = 0.205$, SE = 0.08, $P < 0.01$; Table 5.4). While included in the top models, amount and length of thinned forest edge within each buffer had no effect on predicted abundance in 2020 ($\beta = -0.116$, SE = 0.10, $P > 0.05$) but increasing amounts ($\beta = -0.267$, SE = 0.10, $P < 0.01$) and length of edge ($\beta = -0.258$, SE = 0.10, $P < 0.05$) within the 750 m buffer had a negative effect on estimated abundance in 2021.

The probability of detection was 0.499 in both years (2020 CI: 0.498–0.501 and 2021 CI: 0.498–0.500). The percent of lunar illumination did not affect detection probability for either season and was not included in any retained models. The amount of cloud cover, as determined by an increasing sky percent value, was included in competing models in both years, but did not influence detectability ($\beta = -0.003$, SE = 0.003, $P = 0.27$ and $\beta = -0.004$, SE = 0.003, $P = 0.17$, respectively; Tables 5.3 and 5.4). The proportion of managed pine around each survey point, grouped by age class, was different between survey years, specifically with a greater percent coverage of pine 0–5 years since stand initiation around each point in 2020 and higher percent coverage of pine 16–20 years since stand initiation in 2021 (Figure 5.7). Estimated abundance using the N-mixture models, within the areas surveyed, as a weighted average of all models within a ΔAICc of 2, was 1579 individuals (CI: 1521.87–1636.90; 1 male per 0.79 hectares) in 2020 and 1020 individuals (CI: 771.93–1267.48; 1 male per 2.66 hectares) in 2021.

Capture Methods and Captured CWWI Ages and Measurements

I captured four CWWI (three males and one female) during May and June 2021 (Table 5.5). All captured males had molted their rectrices as indicated by the white patches on the outer feathers. Two of the captured males had molt limits in the primaries; CWWI frequently begin to molt earlier in the summer than most other passerines and near-passerines depending on food availability, with a full molt taking 3–4 months on average to complete (Figure 5.8; Rohwer 1971, Pyle 1997). Because of the molt limits and plumage coloration, I was able to age these two birds as second year (SY) individuals. The third male did not have any molt limits and was comparatively greyer, indicating that it was an after second year (ASY) bird. I determined the captured female to be a SY bird, as indicated by the molt limit in the greater coverts and cinnamon coloration of the flight feathers. The female weighed 112 g and the average male mass was 118.7 g. This is consistent with previous studies (Straight and Cooper 2020).

I captured all individuals using the modified mist net, decoy, and playback method; no other method resulted in successful capture of any CWWI (Table 5.6). All four individuals were captured under different night conditions (Table 5.7). Cloud cover ranged from 20–100%, with an average temperature of 25.78º C. No wind speed was detected during any of the successful trapping events. However, there was greater than 95% lunar illumination on 3 or 4 successful capture nights. My mist net modification was highly successful in capturing CWWI as it reduced the likelihood of these birds bouncing out of the mist net. Using the decoy, speaker, and modified 60 mm mist net, I captured 4 of 5 CWWIs that made contact with the net. There was only one occurrence where a Chuck-will's-widow hit the net and bounced out; this bird hit net close to the poles in the second shelf where net tension remained high. Use of only the decoy, mist net, and playback resulted in CWWI bouncing out any time contact with the net was made, while CWWI did not directly approach the mist net when only playback was used. CWWI did not approach the bow net. My net setup also captured a Northern Bobwhite (*Colinus virginianus*).

During 18 of 23 trap nights, I observed at least two individuals responding by returning calls, moving closer to the net, and sitting in nearby tree branches. In many of these events, no birds directly approached the net nor the speaker. But when I used the decoy in combination with playback and an unmodified mist net, 100% of individuals bounced out of the mist net, regardless of mesh size. The combined approach of decoy and playback provoked responses by both males (identified by presence of white patches on retrices) and females. Typical responses included grunting and bill snapping, fanning of the tail, tail wagging, and wing-clapping. Although generally effective at attracting individuals, on three occasions I determined Chuck-will's-widows were present, but not approaching the net, suggesting that there are additional factors influencing responses. When nets were erected over grass or were surrounded by mid- to late-rotation pine stands, individuals never approached and instead vocalized from branches directly above the speaker.

Blood Isotope Values

Average $\delta^{13}C$ and $\delta^{15}N$ from the blood samples were -25.72 and 4.94, respectively. When compared to loblolly pine, switchgrass, and *Rubus* spp. samples that were previously collected in the same working forest, results indicated that CWWI captured prey that could be attributed to C3 plants such as pine and shrubs (Figure 5.9).

Discussion

Through my study I found local abundances of CWWI are supported by the availability of multiple pine forest conditions within the larger landscape. The amount of open pine conditions, specifically created by recently replanted forest stands in the early stages of regrowth, are clearly documented to have a positive relationship on local CWWI abundances. These findings are important as, despite ongoing efforts to create appropriate foraging and nesting conditions for many avian species, the effects of active forest management on nightjars remains largely unknown (Straight and Cooper 2020). It is well established that aerial insectivore populations are declining (Sauer et al. 2017), but for most of these species, few focused efforts to identify specific local causes have been undertaken. Recent research has indicated that there are breeding habitat associations for some of the more commonly studied species, such as the EWPW, specifically including shrublands and early successional forests, especially those with mixed oak (*Quercus* spp.), beech (*Fagus* spp.), and pine (*Pinus* spp.) and with open edges and uncrowded trees (Tyler 1940, Mills 1986, Wilson and Watts 2008, Hunt 2013, Tozer et al. 2014, Farrell et al. 2017).

Even though breeding habitat associations remain ambiguous for much of the CWWI range, my surveys and corresponding results indicate that managed pine forests with these described conditions likely maintain high breeding abundance levels. The amount of early successional forest was identified to be important in predicting CWWI abundance at multiple spatial scales. It is possible that this is explained by examining suspected CWWI foraging locations, which are thought to be in forest openings, pastures, and low-lying scrub (Straight and Cooper 2020). Like for EWPW, these recently cleared forest stands may serve as an alternative to natural early successional vegetation conditions (Tozer et al. 2014, Farrell et al. 2017), and is consistent with previous research investigating the preference of aerial insectivores for clearcut and open matrices such as logging roads and loading decks within my study system (Wilson and Watts 2008, Farrell et al. 2017, Vala et al. 2020).

I expected that because CWWI are aerial insectivores and visual foragers the presence of recently cleared forest stands within the relatively continuously forested landscape would positively influence occurrence, and similarly, abundance. Within the large, forested landscape, the mosaic style presence of early successional forest conditions and size of clearcut likely create opportunities for maintaining high CWWI abundances throughout the area and not just in isolated patches. While I did not evaluate nest site selection in this study, proximity of a variety of forest stand ages, specifically of clearcuts and early successional stands potentially as foraging sites to thinned, open canopy forests likely as nesting sites, likely allowed for detections of CWWI at every survey point.

Previous research has acknowledged the possible benefit of recently thinned stands and open forest conditions, created by mechanical thinning and decreasing basal area, to both CWWI and EWPW (Wilson and Watts 2008, Spiller and King 2021, Thompson et al. 2022). Given the pine stand age at which most CWWI were detected during my study, my findings support this conclusion. Interestingly, the average age of pine stand where individuals were detected was 17.31 years, with one standard deviation of 15–19 years. This range encompasses both thinned and unthinned stands, and while a large portion of pine stands at 17 years are generally thinned within my study system, most individuals were not detected in thinned stands.

While I did not survey from the middle of each forest stand, CWWI were commonly detected near forest edges. This may be an artifact of survey design, but it could indicate CWWI use the forest edges where light and proximity to open roads and the surrounding matrix and stands is highest. CWWI are known to forage in open conditions, such as open fields and pastures, and within my study clearcuts and roads may be the equivalent. However, models showed a negative relationship to increasing forest edge lengths and area within this system. In a prior study, forest edges were found to support EWPW, but only with increasing numbers of moths (Souza-Cole et al. 2022). However, unlike previous studies (e.g., Mills 1986, Vilella 1995, Reino et al. 2015, Evens et al. 2020), lunar illumination was not shown to affect detection ability within my study area. This was unexpected given the previous research, but with high numbers of detections and predicted abundance, it is likely that CWWI were continuously vocal, with a consistent need for territorial calls within the study system.

Curiously, even given the detection locations, I observed a negative relationship with the amount of later stage pine within the landscape. Examining the landscape leads to a discovery that fewer of the managed pine stands were in the early stages of the rotational process, resulting in more pine forests above 16 years of age. Hence, the surveyed areas generally consisted more heavily of later stage, thinned pine stands with closing canopies. This indicates that, given model results, these open, young forests are the main driver of CWWI abundance within this system and can be considered as the limiting factor to CWWI occupancy within heavily forested areas.

The high number of detections in older forest stands could also be a product of my chosen survey sites. But in general, there was a higher percentage of cover of later rotation forests (15–30 years post planting) across the landscape during both survey years, leading to the assumption that forests of those ages must provide adequate resources, whether food, shelter, or nesting sites, in order for occupancy in these stands to occur. Survey points were structured along drivable roads, but often multiple points in a row were surveyed during the same time, leading to an overlap in habitat covariate values at the largest buffered distance. In a similar fashion, multiple models were found to be significant, but no one model greatly explained my observed detections. More analysis is needed, specifically to include stand adjacency and proximity analyses, both of which could play an important role in determining CWWI occupancy.

As such, some caution is needed when interpreting CWWI habitat preferences as little is known about nest site selection and availability of food sources created by the different forest conditions within my study site. Unlike EWPW, previous research has found that CWWI may not be as dependent on Lepidopteran prey, with up to 60 % of their diet consisting of beetles in the Scarabaeidae family (Bent 1940). This could drive the observed differences between the observed CWWI blood $\delta^{13}C$ and $\delta^{15}N$ isotope values compared to collected Purple Martin (*Progne subis*; hereinafter PUMA) samples from the same field site (unpublished data). PUMA commonly consume a wider variety of prey than CWWI, capturing a higher proportion of flies, true bugs, and bees and wasps, as well as a variety of Lepidopteran butterflies and moths (Brown et al. 2021). It can be inferred that as CWWI are visual foragers, the openness of recently cleared stands provides the appropriate opportunities for foraging. This suggests that within my survey area, and by association the entirety of the managed landscape, CWWI may preferentially select stands with high amounts of open foraging areas available nearby.

Combining these ideas, CWWI occupancy is likely dependent on both availability of young open pine and mid-rotation thinned stands. This could indicate that CWWI are a species that is influenced by landscape configuration and composition. CWWI are likely a species that depends on resources not contained within one habitat type and must extend their movements into multiple forest conditions. As such, space use within this managed landscape may not only depend on the quality of an individual stand, but also on the composition of the surrounding forested landscape (Vala et al. 2020). Thinned stands may provide appropriate nesting sites while clearcuts and young open canopy stands may provide more foraging opportunities. Since CWWI are visual foragers, open forest conditions, especially during nights with high lunar illumination, may enhance their ability to detect prey.

Detectability of this species should be taken into account for any further studies. Number of detections per survey point did not significantly differ by date throughout the sampling period. During my surveys, I detected 259 of the 295 individuals within the first 10 minutes of the survey period. While I did not specifically note continuation of call, most birds ceased calling within 5–10 minutes of first detection. Additional surveys for the concurrent CWWI banding project further solidified this finding; most individuals stopped responding to playback within 1 hour of sunset. Current Nightjar Survey Network protocols dictate that surveys should begin 30 minutes after sunset and continue for 6 minutes per 10 survey stops along a specified route (Nightjar Survey Network), which gives a total survey time of 60 minutes not including travel time between points. Given my surveys, where the detection of CWWI was significantly lower later in towards the end of each individual survey period, I recommend that future surveys should be designed to account for this variation.

Trapping CWWI within this study area led to a variety of difficulties, but eventually successful captures. The use of a decoy and conspecific song playback proved successful for attracting CWWI to the trapping site, but their fast speed combined with larger size led to consistent bouncing upon impacting both 30 mm and 60 mm mist nets. The latex ring modification allowed me to keep the net out in the open while having the added flexibility necessary to circumvent previous issues when birds struck nets with high speeds and force. I had considered placing the mist net within areas of denser vegetation to slow their momentum, but this method became impractical due to vegetation densities and bird behaviors. When nets were close to adjacent woodlots, I observed CWWI singing to and duetting with the decoy while perched on tree branches or snags directly above the nets. These birds would never approach the decoy directly.

Additionally, keeping the decoy visible on open, rocky, or barren substrates emerged as an essential component to attracting CWWI to the site compared to grassy or forested substrate. Two separate capture events resulted from the birds sitting on the ground adjacent to the net where they sang and/or displayed at the decoy. The observed displaying behaviors were consistent with courtship and territorial displays previously documented (Bent 1940, Mengel and Jenkinson 1971, Holyoak 2001). When these birds eventually began to fly off and reposition themselves, they were caught in the bottom two shelves of the mist net. Ensuring that the decoy is properly positioned on preferred substrate not only attracts the CWWI to the nets but also to the decoy itself, thus increasing the number of capture opportunities. I strongly recommend, based off observations made during this study, that grassy or otherwise covered ground not be considered for decoy placement due to the hesitation of CWWI to land directly on or near these obstacles.

Net placement within the managed pine forest was determined to be important to the overall success of capture efforts. Successful capture events incorporated nets that were placed on open gravel or clay substrates, typically access or logging roads adjacent to thinned stands, with the thinned stand to the west of the net allowing for darker conditions quickly as the sun sets behind the stand. The condition of these landscapes can drastically change between years (Connor and Dickson 1997, Wilson and Watts 2000), with the creation of new sites during timber logging and the removal of sites due to increased vegetation growth. As such, there were a limited number of ideal trapping locations within my study area each season. Many of the main roads within the forest were public rights-of-ways and could not be used during the study.

At several locations used for target netting, I attempted to capture multiple individuals on different nights. Multiple individuals typically responded to the conspecific playback, leading to the suspicion that more than one individual could potentially be captured at each location. This high individual response occurrence likely indicated that many of the chosen trapping sites were on or near territory edges, and the presence of an unknown individual (via the conspecific playback) elicited a territorial response from neighboring individuals. Male and female CWWI are plumage dimorphic once a full molt cycle has been completed, as noted by presence of white patches on male rectrices (Pyle 1997), but it is impossible to differentiate one individual from another of the same sex without capture or the placement of external color bands. Consequently, given observed behaviors and the number of initially responding individuals at each capture site, it is highly likely that different individuals responded to each capture event. Even with repeated trapping at previously successful locations, no previously banded birds were captured.

Conclusion and Management Implication

My study demonstrated how maintaining early successional forest conditions, particularly through the use of clearcut harvesting and subsequent replanting, can be beneficial for CWWI. The proximity of recently thinned, mid-rotation stands with open canopies and little ground cover to open forest patches may provide both appropriate nesting and foraging sites. This suggests that managed forests and the conditions created within may serve the same ecological purpose as natural early successional vegetative communities. As such, more attention is needed in these areas as they may provide continual breeding conditions when other potential breeding habitat locations are lost. Given that every site was occupied by at least one CWWI during surveys, the focus should remain on following standards set by the Sustainable Forestry Initiative (2022) that aim to protect species of conservation concern, specifically by continued monitoring and by maintaining or creating both pine-grassland and open canopy forests to support both nesting and foraging requirements for CWWI.

Landscapes with high proportions of forest stands at either end of the management spectrum (very young or right before harvest) with little variation may restrict foraging and nesting opportunities for CWWI. My research also revealed relatively high abundance estimates for CWWI, compared to the limited number of studies in other geographic locations, in an area that has not previously been surveyed. Much of the southeastern U.S. is managed for timber production, which means that these types of surveys within privately managed landscapes could have serious implications for estimating population size and aid in the conservation of the species. Further studies need to address reproductive success, survivorship, and movement within this landscape to assess the impacts of management actions on CWWI fitness.

Tables and Figures

Table 5.1 Description of covariates used in N-mixture models using Chuck-will's-widow night survey data. Surveys were conducted in May and June of 2020 and 2021 in a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi.

	Covariate	Description	Units
Site			
	AP1_375/750/1125	Amount (scaled) managed pine 1–5 years since initiation, within three buffered distances of survey point	Meters squared
	AP2_375/750/1125	Amount (scaled) managed pine 6–10 years since initiation, within three buffered distances of survey point	Meters squared
	AP3_375/750/1125	Amount (scaled) managed pine 11–15 years since initiation, within three buffered distances of survey point	Meters squared
	AP4_375/750/1125	Amount (scaled) managed pine 16–20 years since initiation, within three buffered distances of survey point	Meters squared
	AP5_375/750/1125	Amount (scaled) managed pine 21–25 years since initiation, within three buffered distances of survey point	Meters squared
	AP6_375/750/1125	Amount (scaled) managed pine 25–30 years since initiation, within three buffered distances of survey point	Meters squared
	EdgeArea375/750/1125	Amount of (scaled) forest edge within 3 buffered distances of survey point	Meters squared
	EdgeLength375/750/1125	Length of (scaled) forest edge within 3 buffered distances of survey point	Meters
	TEdgeArea75/750/1125	Amount (scaled) of thinned forest edge within 3 buffered distances of survey point	Meters squared
	TEdgeLength375/750/1125	Length (scaled) of thinned forest edge within 3 buffered distances of survey point	Meters
	NTEdgeArea375/750/1125	Amount (scaled) of unthinned forest edge within 3 buffered distances of survey point	Meters squared
	NTEdgeLength375/750/1125	Length (scaled) of unthinned forest edge within 3 buffered distances of survey point	Meters
Observation			
	Sky	Estimated cloud cover at start of survey	Percent
	Moon	Lunar illumination	Percent
	Wind	Wind speed	Kilometers/hour
	Temp	Temperature at start of survey	Celsius
	Precip	Daily precipitation	Inches

Table 5.2 The N-mixture models for estimating Chuck-will'-widow abundance, degrees of freedom (*df*), log likelihood (logLik), AICc, ΔAICc, and model weight (W_i), for May and June 2020 and 2021, within managed loblolly pine (*Pinus taeda*) stands in east-central Mississippi.

Year	Model	Covariates	*df*	logLik	AICc	Δ AICc	W_i
2020	Y1_1	AP4_1125 + TEdgeLength1125	4	-85.989	179.98	0.000	0.250
	Y1_2	AP4_1125	3	-87.019	180.04	0.062	0.242
	Y1_3	Sky + AP4_1125 + TEdgeLength1125	5	-85.289	180.58	0.600	0.185
	Y1_4	Sky + AP4_1125	4	-86.446	180.89	0.915	0.158
	Y1_5	AP4_750	3	-87.778	181.56	1.578	0.114
	Y1_6	Null	2	-89.898	183.80	3.819	0.037
2021	Y2_1	Sky + AP1_375 + EdgeArea750	5	-80.926	171.85	0.000	0.472
	Y2_2	Sky + AP1_375 + EdgeLength750	5	-81.008	172.02	0.160	0.435
	Y2_3	Sky + AP1_375	4	-84.395	176.79	4.940	0.040
	Y2_4	AP1_375	3	-85.746	177.49	5.640	0.028
	Y2_5	Sky	3	-86.307	178.61	6.760	0.016
	Y2_6	Null	2	-87.993	179.99	8.130	0.008

Table 5.3 AICc table and estimated abundances for Chuck-will's-widow within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi during May and June 2020 using N-mixture models. Models were considered competing if ΔAICc was less than 2. For each model, covariates are listed with log scale parameter estimates and 95% confidence intervals in parentheses. N = estimated abundance rounded to the nearest whole number. Bolded parameters indicate statistical significance (α = 0.05).

Model	Detection Covariate (estimate, 95% CI)	Site Covariate (estimate, 95% CI)	ΔAICc	N (95% CI)
1		**AP4_1125 (-0.230 [-0.424, -0.037])** TEdgeLength1125 (-0.116 [-0.278, 0.045])	0.000	1592 (1235, 1973)
2		**AP4_1125 (-0.219 [-0.408, -0.029])**	0.062	1596 (1240, 1976)
3	Sky (-0.003 [-0.009, 0.002])	**AP4_1125 (-0.279 [-0.493, -0.066])** TEdgeLength1125 (-0.003 [-0.280, 0.038])	0.600	1532 (1184, 1904)
4	Sky (-0.003 [-0.009, 0.003])	**AP4_1125 (-0.262 [-0.471, -0.054])**	0.915	1532 (1183, 1901)
5		**AP4_750** (**-0.176 [-0.349, -0.003])**	1.578	1659 (1292, 2050)

Table 5.4 AICc table and estimated abundances for Chuck-will's-widow within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi during May and June 2021 using N-mixture models. Models were considered competing if ΔAICc was less than 2. For each model, covariates are listed with log scale parameter estimates and 95% confidence intervals in parentheses. N = estimated abundance rounded to the nearest whole number. Bolded parameters indicate statistical significance (α = 0.05).

Model	Detection Covariate (estimate, 95% CI)	Site Covariate (estimate, 95% CI)	ΔAICc	N (95% CI)
1	Sky (-0.004 [-0.009, 0.001])	**AP1_375 (0.205 [0.058, 0.352])** **EdgeArea750 (-0.267 [-0.469, -0.065])**	0.000	1001 (733, 1291)
2	Sky (-0.004 [-0.009, 0.001])	**AP1_375 (0.201 [0.053, 0.349])** **EdgeLength750 (-0.258 [-0.460, 0.002])**	1.578	1040 (764, 1336)

Table 5.5 Collected measurements, sex, and age for Chuck-will's-widow captured in a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi.

Date Captured	Sex[a]	Age[b]	Wing Chord[c]	Mass[d]	Fat Score[e]	Sex Characteristic[f]
May 24, 2021	F	SY	215	112	1	BP
May 25, 2021	M	ASY	211	112	1	No CP Observed
May 27, 2021	M	SY	219	117	0	No CP Observed
June 4, 2021	M	SY	220	127	0	No CP Observed

[a] Sex abbreviations: F - female, M - male

[b] Age abbreviations: SY - second year, ASY - after second year

[c] Wing Chord is measured in millimeters

[d] Mass is measured in grams

[e] Fat Score is scored 0–5, with 0 meaning no fat observed in the furculum and 5 meaning an overflowing amount of fat in the furculum and often around the base of the tail and legs (modified from North American Banding Council 2001 and Redfern and Clark 2001)

[f] Sex Characteristics abbreviations: BP - brood patch, CP - cloacal protuberance

Table 5.6 Chuck-will’s-widow trapping methods used, and number of attempts and captures per method. From May–June 2019–2021 within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi.

Trapping Method	Number of Attempts	Number of Approaches	Number of Bounces	Number of Successes
Single 30 mm mist net with playback	6	0	0	0
Single 30 mm mist net with playback and decoy	2	2	2	0
“V” configuration 30 mm mist nets with playback and decoy	2	1	1	0
“V” configuration 30 mm and 60 mm mist net with playback and decoy	2	1	1	0
Bownet with playback and decoy	1	0	NA	0
Single 60 mm mist net with playback and decoy	2	2	2	0
Single 60 mm mist net with latex rings, playback, and decoy	8	5	1	4

Table 5.7 Weather conditions during successful Chuck-will's-widow trapping events, May–June 2021 within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi.

Date	Longitude	Latitude	Capture Time	Temp[a]	Moon[b]	Wind[c]	Cloud[d]
5/24/2021	-88.515097	32.802028	20:30	25	95	0	20
5/25/2021	-88.515097	32.802028	20:20	26.4	97	0	65
5/27/2021	-88.529325	32.801822	20:40	27.7	99	0	80
6/4/2021	-88.536203	32.801483	20:50	24	27	0	100

[a] Temperature in degree Celsius
[b] Lunar illumination in percent
[c] Measurable wind speed in kilometers per hour
[d] Cloud cover in percent

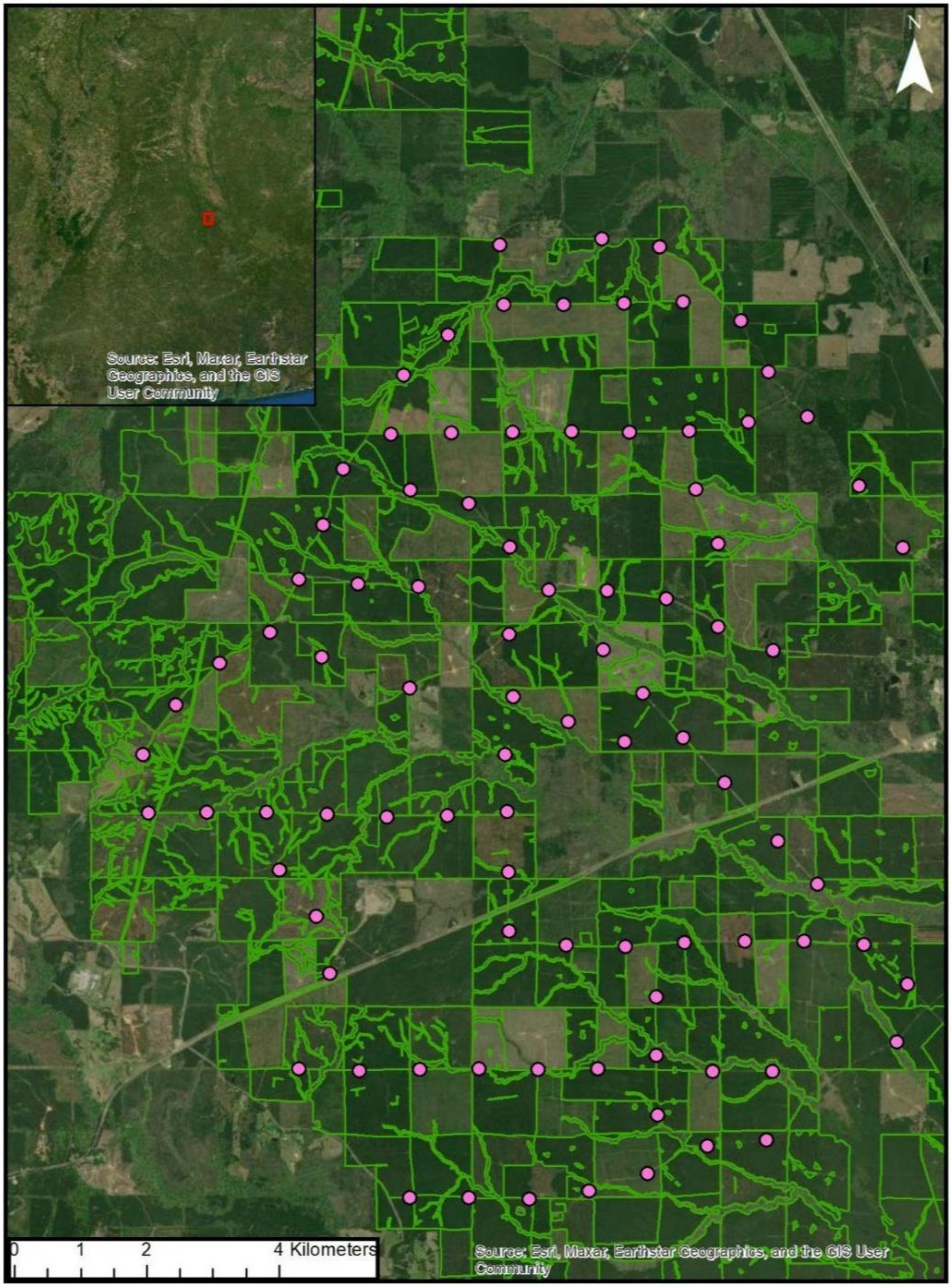

Figure 5.1 Placement of Chuck-will's-widow night survey points within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi, as indicated by pink circles. Weyerhaeuser Company managed forest stands highlighted in light green.

Figure 5.2 Photograph showing the most effective net and decoy placement for capturing Chuck-will's-widow within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. Note the thinned stand with relatively little ground vegetation and an open canopy next to a young, recently planted stand, with a clay and gravel opening and little vegetation in between.

Figure 5.3 Image of fabricated Chuck-will's-widow decoy. Plastic 3-D model overlaid with clay, and painted brown. Model is mostly to scale (length is accurate while head is slightly smaller than life size).

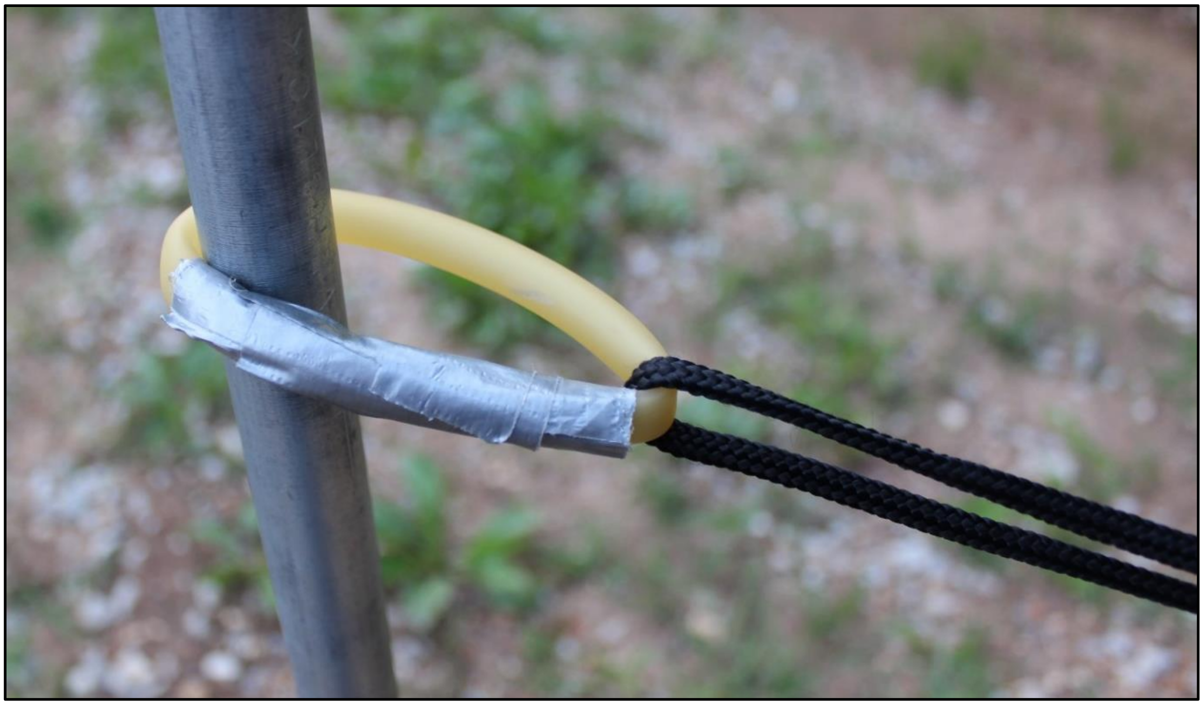

Figure 5.4 Latex hose creating a new trammel loop for the mist net. I used ¼ inch latex hose, cut into 7-inch pieces, to create each loop. Loops were duct taped together to allow for adjustment in the field.

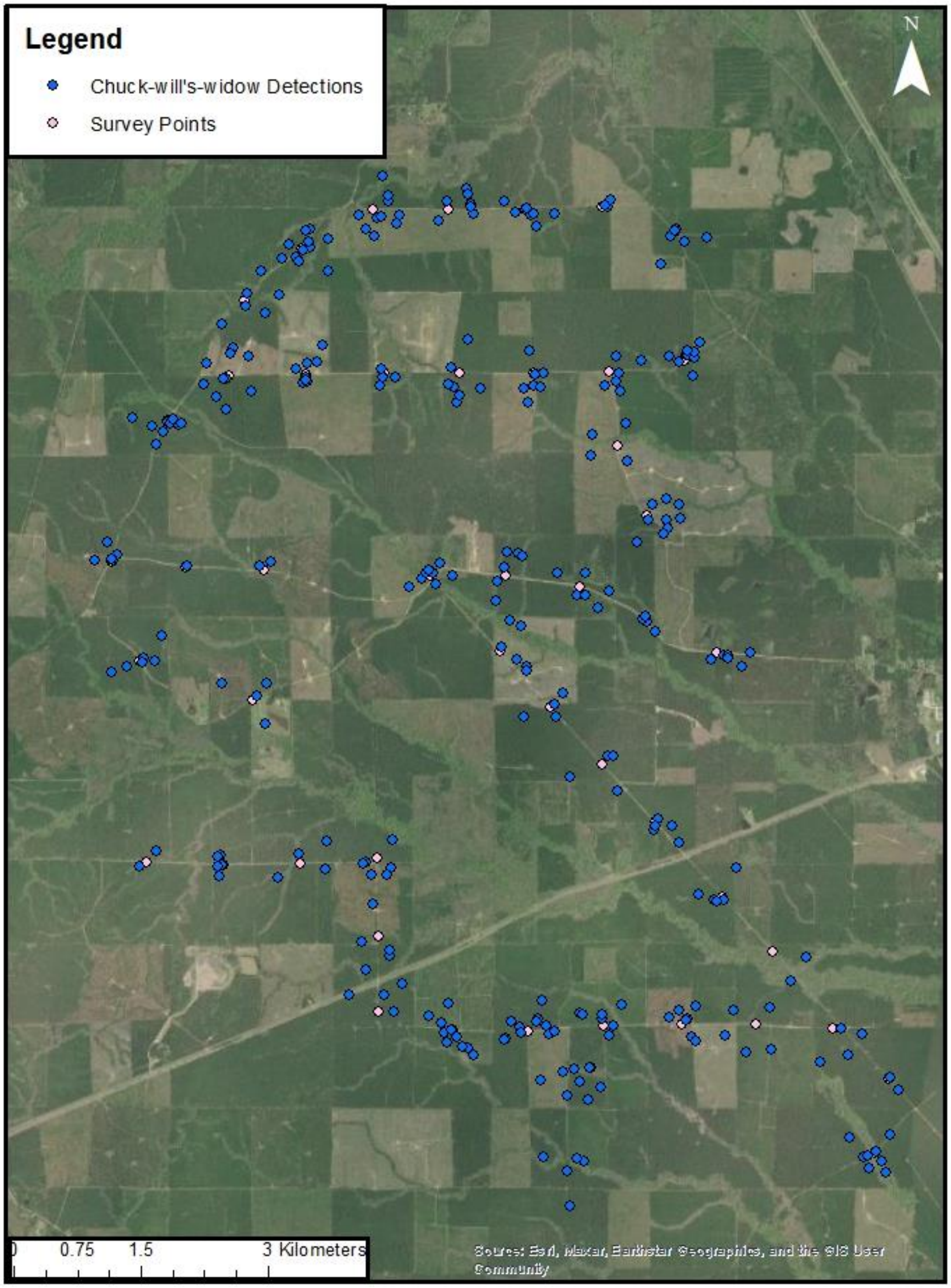

Figure 5.5 Chuck-will's-widow detections during night surveys within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi May–June 2020–2021. Survey points may be overlayed with detection points.

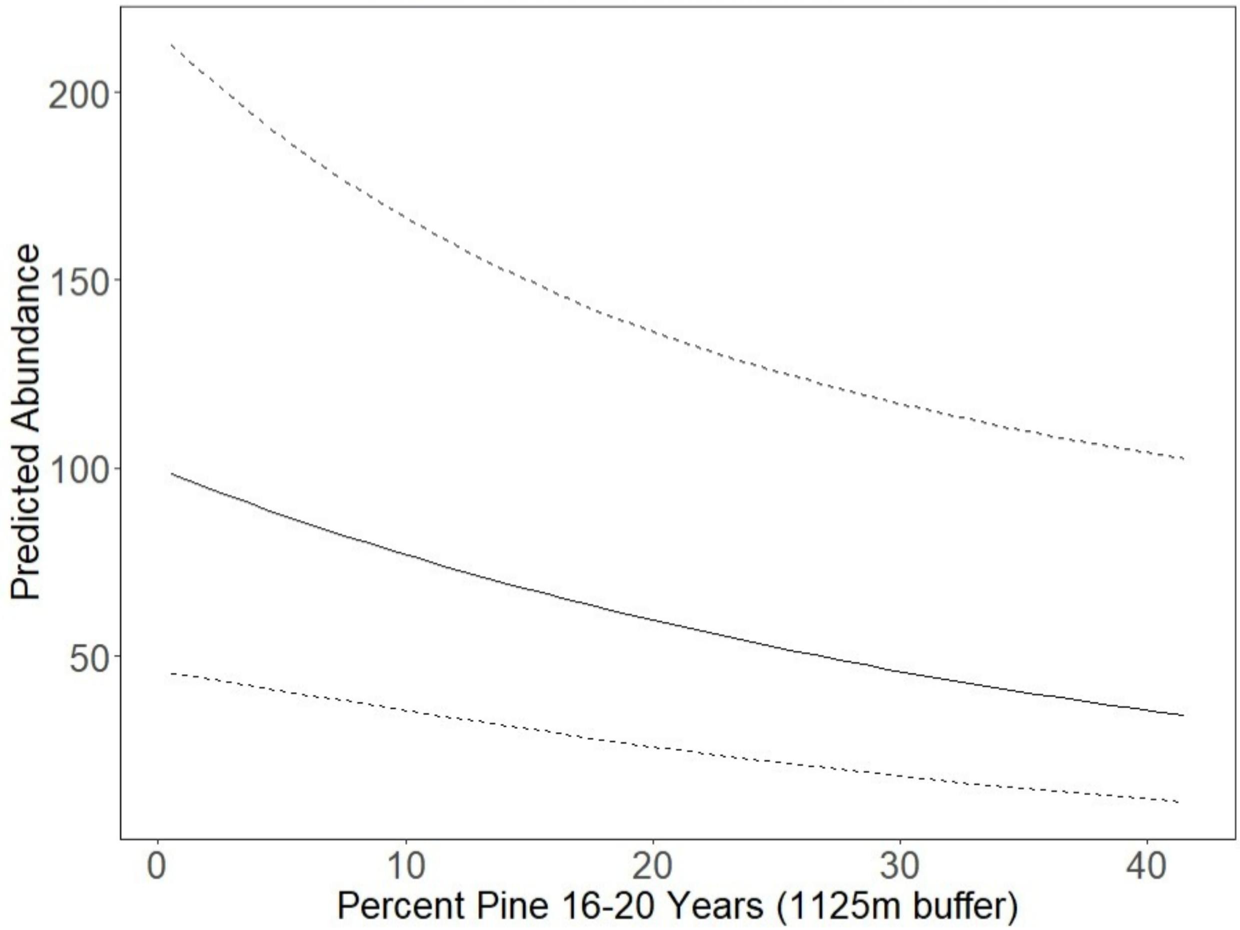

Figure 5.6 Predicted effects of percent of managed loblolly pine 16–20 years since stand initiation on Chuck-will’s-widow abundance estimates from surveys conducted in 2020 within a managed loblolly pine (*Pinus taeda*) forest in east-central Mississippi. Dashed lines represent the smoothed 95% confidence interval..

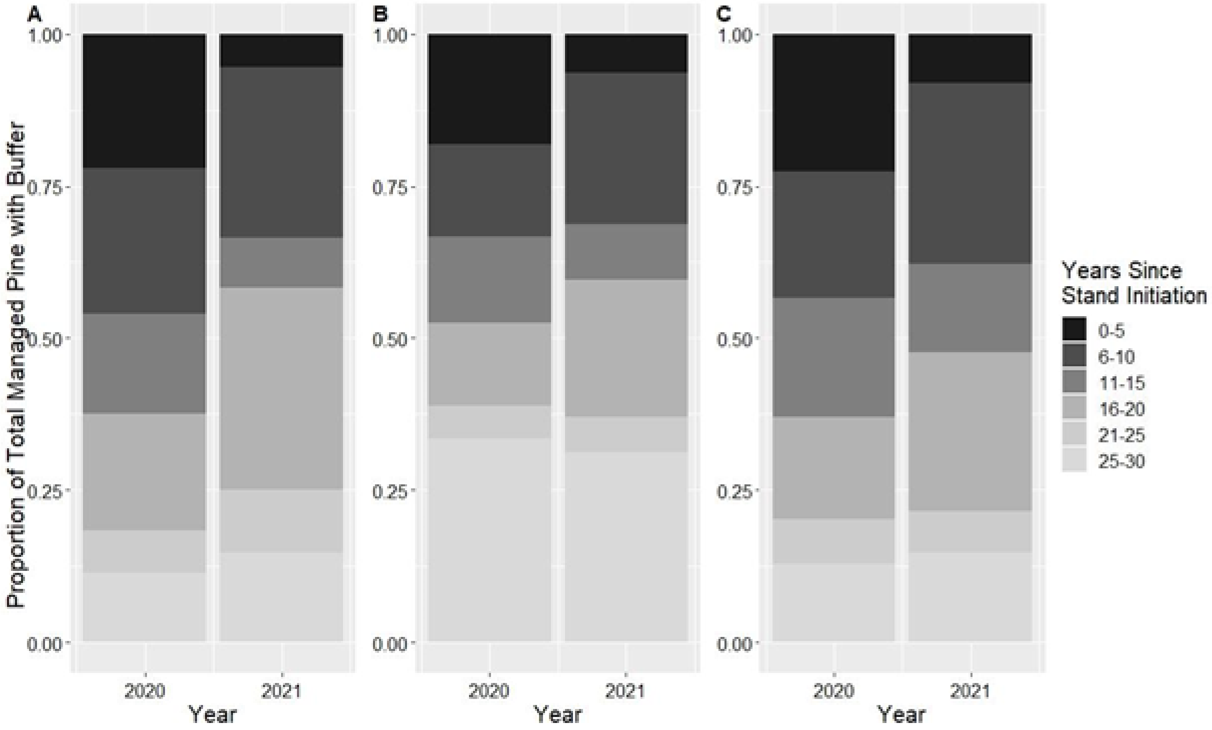

Figure 5.7 Proportion of managed loblolly pine (*Pinus taeda*) in each age class within (a) 375 m, (b) 750 m, and (c) 1125 m buffers, from within the total percent of managed loblolly pine, around Chuck-will's-widow survey points in May and June 2020–2021. Surveys conducted within a managed loblolly pine forest in east-central Mississippi.

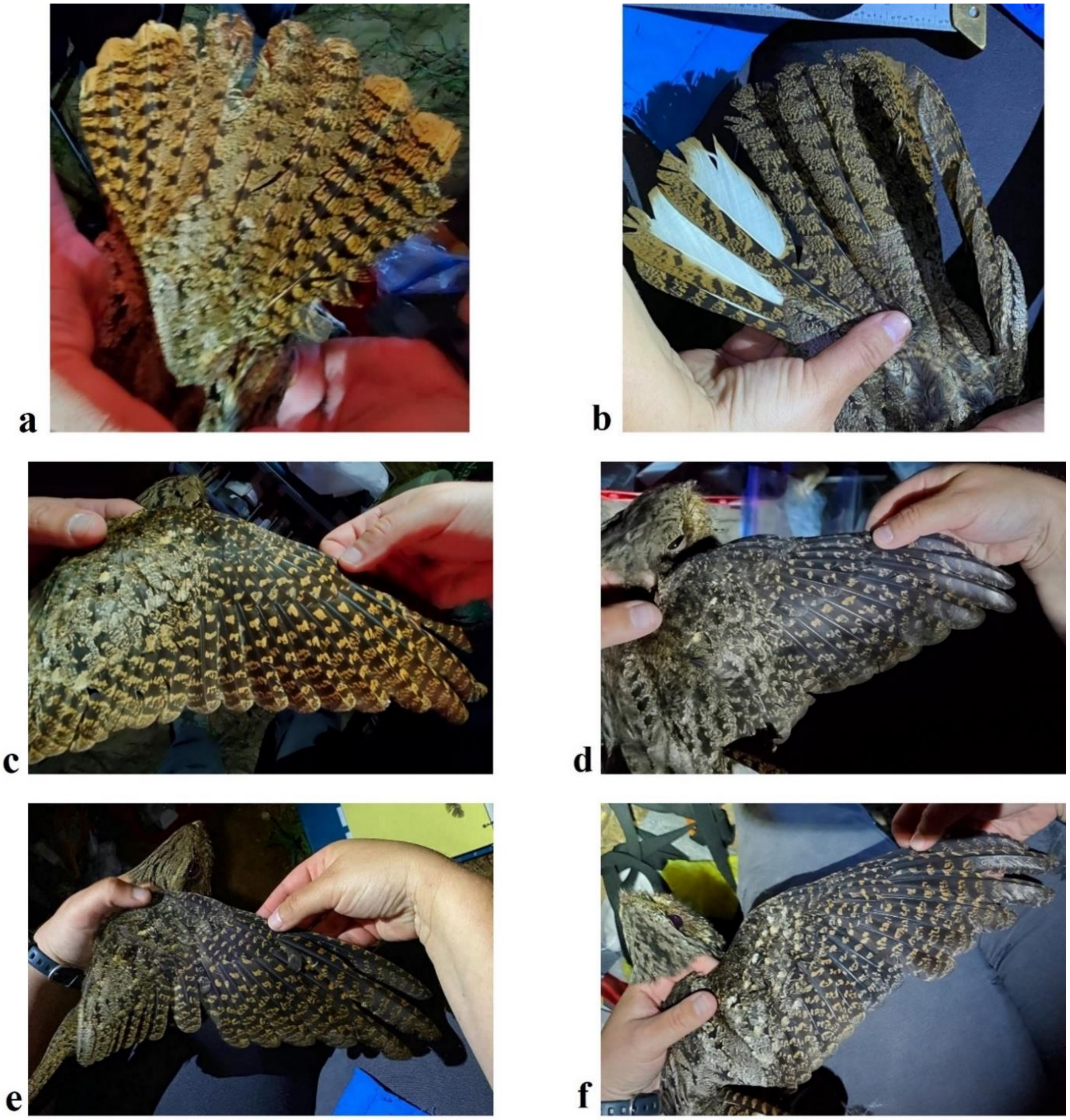

Figure 5.8 Variations in Chuck-will’s-widow tail (rectrices) and wing (flight feather) plumage. Image (a) is a female Chuck-will’s-widow, (b) is a male Chuck-will’s-widow (note the white patches on the outer rectrices, indicating either an older male or a SY male that has completed a partial molt), (c) is a SY female Chuck-will’s-widow, (d) is an ASY male Chuck-will’s-widow, (e) is an SY male Chuck-will’s-widow with an active symmetric molt, and (f) is an SY male Chuck-will’s-widow with a symmetric molt.

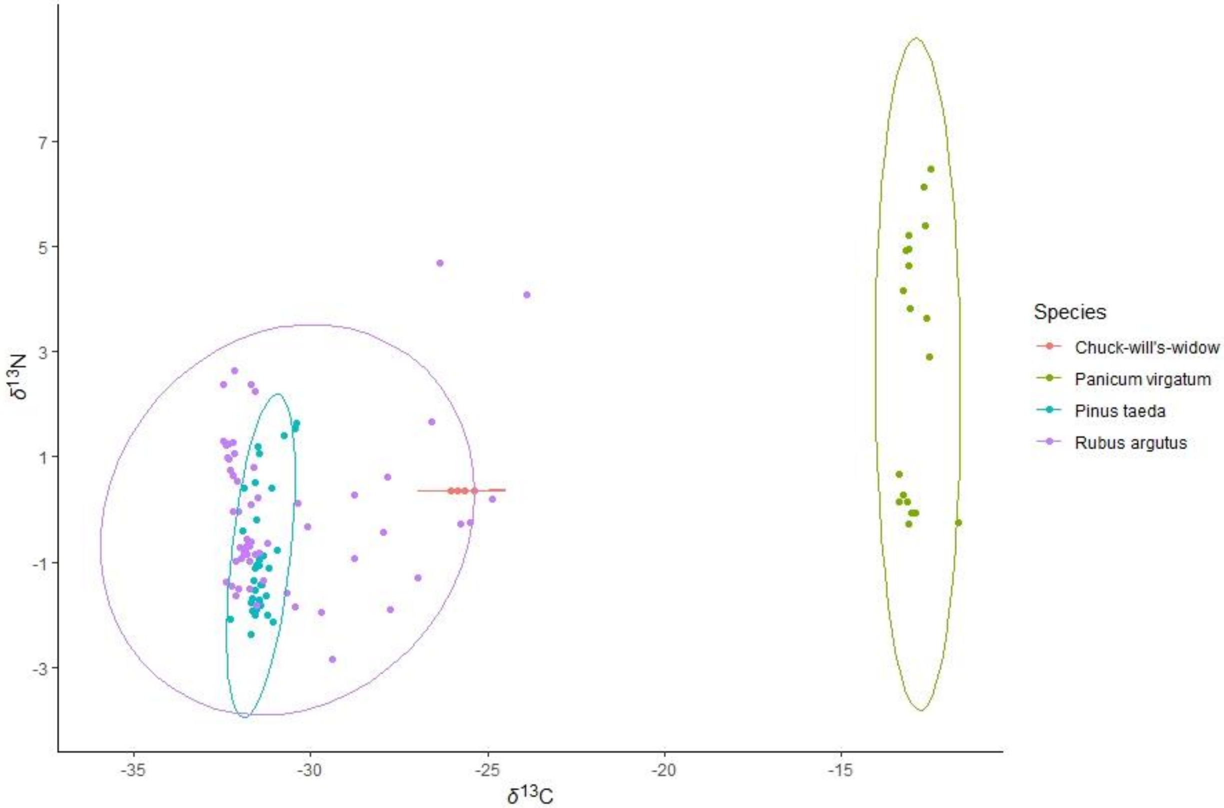

Figure 5.9 Comparison of δ^{13}C and δ^{15}N values from Chuck-will's-widow blood samples (captured 2021), switchgrass (*Panicum virgatum*) blade samples, loblolly pine (*Pinus taeda*) needle samples, and blackberry (*Rubus argutus*) leaf samples collected within managed loblolly stands planted in 2011 in east-central Mississippi. Ellipses represent a multivariate normal distribution with a 95% confidence interval for each species.

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