The effects of common forest management practices on community structure in a southern pine forest

By

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Planted pine (*Pinus* spp.) comprises nearly 10% of the total land cover in the state of Mississippi. Often, understory structure is limited in this system. Thus, managers use a variety of management practices to improve understory biomass and structure. I assessed the impacts of common forest management practices (canopy reduction, prescribed fire, and selective herbicide application) and their combined effects on aspects of community structure. More specifically, I assessed impacts of disturbance intensity on non-native plant invasions, and evaluated how microscale vegetation characteristics influenced use by white-tailed deer (*Odocoileus virginianus*) and wild turkey (*Meleagris gallopavo*). Combining canopy reductions with prescribed fire, which closely mimicked historical intermediate disturbance intensities in this vegetation type, led to the greatest invasion resistance due to high abundances of native plants. Both deer and turkey increased use in areas with high levels of understory cover. Coupling canopy reductions with prescribed fire created the most favorable conditions for both species.
DEDICATION

This is for those whom have inspired me through their love of the outdoors.

Nothing has shaped my life like the experiences I have had with friends and family while hunting and fishing.
ACKNOWLEDGEMENTS

I want to begin by thanking my family and friends for their support during the past two years. I haven’t been around as much as I had liked, but it seems you all knew just when I needed a phone call. Also, thank you to my fellow graduate students. Our nights at Rosey Baby’s, crawfish boils, and unsuccessful hunting trips kept me sane. To Ashley, thank you so much for the love and support over the past two years. You have made this entire experience more bearable.

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CHAPTER I
INTRODUCTION

Historically, the southeastern United States was dominated by grasslands and open pine savannas maintained by frequent fire. Sources of these fires included lightning and anthropogenic practices (Pyne 1982). However, fire suppression beginning in the late 19th century led to drastic changes in vegetative structure and composition (Mitchell and Duncan 2009). Coupling that with the shift to the plantation pine model to maximize timber production, the changing land use led to widespread habitat degradation for many wildlife species. Today, managers and biologists often recommend using aggressive management practices to restore more natural vegetative communities and improve habitat for desired wildlife species (Edwards et al. 2004, Lashley et al. 2011).

Managers commonly use management techniques such as prescribed fire, herbicide, and canopy reductions in combination to quickly restore degraded plant community structure (Edwards et al. 2004, Chamberlain and Miller 2006, Lashley et al. 2011). Canopy reductions alters the structure and composition of the understory by increasing sunlight penetration to the forest floor stimulating growth of understory vegetation (Morriss 1954, Ford et al. 1993). Without periodic disturbances, such as prescribed fire, woody vegetation encroaches the midstory eventually leading to closed canopy conditions (Jackson et al. 2007, Greenberg et al. 2011).
Prescribed fire has been one of the most widely used management practices in the southeastern United States due to its ability to reduce woody encroachment and create favorable habitat for many wildlife species (Lashley et al. 2015, Harper et al. 2016). Canopy reductions and prescribed fire are often used to increase forage quantity for white-tailed deer (*Odocoileus virginianus*, Demarais et al. 2000, Edwards et al. 2004, Mixon et al. 2009, Iglay et al. 2010) and improve structure for wild turkey (*Meleagris gallopavo*; Pack et al. 1988). However, controlling hardwood encroachment with fire alone can be very time consuming, so managers often opt to use a more aggressive method of hardwood control.

Combining canopy reductions with prescribed fire and selective herbicide application is a management technique recently used by managers to expedite conversion to an herbaceous dominated understory (Edwards et al. 2004). Research has demonstrated the importance of herbaceous dominated understories for both white-tailed deer (Edwards et al. 2004) and wild turkey (Porter 1992, Godfrey and Norman 1999). However, other studies have demonstrated the lack of understory structure for up to 3 years post treatment (McCord et al. 2014). Further research is needed to determine the efficacy of this treatment at different temporal scales.

I addressed gaps in the literature by examining effects of common forest management practices on aspects of community structure. In Chapter II, I assessed the effects of common forest management practices (canopy reduction, prescribed fire, and selective herbicide application) and their combined effects on non-native plant invasions. To better understand the mechanism driving invasion resistance, I examined the relationship between non-native and native species richness and abundance. In Chapter
III, I examined how vegetative characteristics affects habitat use by white-tailed deer and wild turkey at the micro scale. In this chapter, the management practices were used to create a wide range of vegetative conditions present on the study site.
Literature Cited


CHAPTER II
NATIVE SPECIES ABUNDANCE BUFFERS NON-NATIVE PLANT INVASIBILITY FOLLOWING INTERMEDIATE DISTURBANCE

Introduction

The intermediate disturbance hypothesis (IDH) is a fundamental ecological hypothesis used to explain the relationship between disturbance intensity and diversity (Connell 1978, Hobbs et al. 1984, Hobbs and Huenneke 1992, Townsend & Scarsbrook 1997, Flöder and Summer 1999, Molino and Sabatier 2001). The IDH predicts species diversity is maximized at an intermediate disturbance intensity. Herein, I use the definition of disturbance proposed by White and Pickett (1985) - any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment. The IDH hinges on ruderal species being most competitive following frequent disturbance and long-lived species being most competitive following infrequent disturbances. However, a mixture of ruderal, long-lived, and intermediate species can survive in an intermediate disturbance regime facilitating relatively high plant diversity (Hobbs and Huenneke 1992).

A hypothesis linked to the IDH is diverse plant communities more effectively utilize niche space resisting invasion from non-native plants, known as the biotic resistance hypothesis (BRH, MacArthur 1955, Elton 1958). Data from small-scale experiments have demonstrated this pattern (Kennedy et al. 2002, Fargione and Tilman 2006).
however, large-scale observations indicate a positive relationship between native and non-native plant diversity (Levine and D’Antonio 1999, Stohlgren et al. 1999, Fridley et al. 2004, Belote et al. 2008). This “invasion paradox” is commonly attributed to differences in the scale of studies where larger areas exhibit greater spatial heterogeneity than smaller areas, thereby providing more niche space for nonnative species to invade (Chisholm. 2009, Powell et al. 2013). However, others contend that a positive relationship between native and non-native species richness only emerges without strong invading species because strong invaders begin displacing other species (Ortega and Pearson 2005).

Understanding how disturbance affects non-native invasions in various ecosystems is important to develop comprehensive invasion mitigation frameworks (Fridley et al. 2007). Often, disturbance creates a dilemma because it is necessary for restoration and maintenance of many ecosystems, but disturbance frequency or intensity may come with a tradeoff of increasing invasibility (Hobbs and Huenneke 1992). For example, in the Southeastern United States, ecosystems were traditionally maintained by frequent disturbance such as wildfires ignited by lightning and indigenous peoples (Pyne 1982). In these systems, it is widely accepted that conservation depends on the use of prescribed fire and other disturbances to mimic historical disturbances (Masters et al. 2003). However, there is a lack of empirical evidence describing the relationship between commonly implemented disturbance regimes on the nonnative plant invasibility of southern pine forest communities.

In my study, I addressed this issue by (1) examining the relationship between abundance of native and non-native species; (2) evaluating the relationship between
native and non-native species richness; and (3) evaluating how a gradient of disturbance intensities (canopy reduction, canopy reduction with fire, and canopy reduction with herbicide and fire) affect the success of non-native invasions. I tested competing hypotheses related to the “invasion paradox” and determined the efficacy of using the IDH to predict invasibility. Additionally, I evaluated how native species abundance affected non-native invasibility. I predicted native and non-native species richness and abundances would have negative relationships, and an intermediate disturbance regime would lead to the greatest invasion resistance (Figure 2.1).

**Materials and Methods**

**Study Site**

My study was conducted at Andrews Forestry and Wildlife Laboratory in Oktibbeha County, Mississippi. It is a 200-ha, 27-year-old loblolly pine (*Pinus taeda*) forest (Figure 2.2). The study area was within the flatwoods topographic region with gently sloping terrain at an elevation ranging from 90 to 101 m. The dominant soil type present on the study area was the Falkner silt loam with zero to five percent slope and the average annual rainfall is ~140 cm (U.S. Climate Data).

Combinations of canopy reduction, herbicide and fire were implemented within stands to represent a gradient in disturbance intensity. In 2014, 18 stands ranging from 5-22 ha in size were thinned to target basal areas (BA) of 9, 14, or 18 m2/ha. However, mechanical canopy reduction was inherently variable at the stand level producing a gradient in basal area ranging from 5-30 m2/ha at the plot level within those stands (See below for plot description). In one stand of each thinning intensity, imazapyr and metsulfuron-methyl were applied to remove understory and midstory vegetation in
October 2014. In May 2015, prescribed fire was implemented in the herbicide stands and 1 stand of each thinning intensity (6 total).

**Field Sampling**

The study area was systematically divided into a 200-m sampling grid with permanent sampling plots occurring at intersections of the grid lines (n=81; Figure 2.2). In June and August of 2016, 3 30-m point-intercept transects were conducted at each sampling location to measure abundance of native vs. non-native plants and species richness. All understory plant species intercepting the transect line at 1-meter intervals were tallied.

**Data Analysis**

First, I examined the relationship between native and non-native species richness across the study area and within each treatment using Pearson’s correlation. Second, I used the “betareg” package (Cribari-Neto and Zeileis 2010) in Program R (Version 3.3.1, R Development Core Team, 2016) to conduct a beta regression examining the interaction between composition by native and non-native species (i.e., does non-native proportional coverage increase with decreasing coverage by native species?). Beta regression was used to account for proportional data not including 0 and 1. I calculated native and non-native abundance using percent cover from the transect data at each sampling location and performed multiple regression-based analyses. Finally, I conducted two competing beta regression analyses. Both models compared cover by non-natives across treatments with the only difference being the addition of basal area as a predictor. I compared Akaike’s information criterion (AIC) scores to determine the best fit model. There was little
difference between models (i.e., $\Delta$AIC < 2), so I chose the most parsimonious model: percent cover of non-natives as a function of treatment (canopy reduction, canopy reduction + fire, and canopy reduction + herbicide & fire).

**Results**

**Native and non-native species richness**

Using Pearson’s correlation, I observed a positive relationship between native and non-native species richness across treatments in both June ($p<0.01$, DF=76, $r=0.30$) and August sampling periods ($p<0.01$, DF=76, $r=0.29$) (Figure 2.3). Within treatments, I also observed a positive trend with the strongest relationship occurring in the canopy reduction treatment during June ($p<0.001$, DF =39, $r=0.65$) and August ($p=0.0034$, DF=39, $r=0.45$) (Figure 2.4).

**Native and non-native species abundance**

I conducted a beta regression to determine that native and non-native abundance were not related in the June sampling period (Psuedo $r^2 = 0.0122$, $p = 0.3547$). However, in August, native and nonnative species abundance was negatively related (Psuedo $r^2 =0.0974$, $p = 0.0070$) (Figure 2.5).

**Disturbance Intensity**

Using the most parsimonious model, I observed non-native species abundance was greatest in the canopy reduction treatment which was the least intense disturbance during June, but it was similar along the gradient of greater intensities (Table 2.1). However, in August, non-native species abundance was lowest in the intermediate disturbance intensity (Table 2.1).
Discussion

Similar to other studies (e.g., Levine and D’Antonio 1999, Stohlgren et al. 1999, Stohlgren 2002, Fridley et al. 2004, Belote et al. 2008), native and non-native species richness was positively correlated in this study — a pattern known as the biotic acceptance hypothesis (“the rich get richer”, Stohlgren et al. 2003). The BRH predicts a negative relationship between native and non-native species richness; however, my data reject that hypothesis because non-native and native species richness responded similarly to disturbance intensity (Naeem et al. 2000). Nevertheless, evaluating invasion success based on species richness relationships may not be accurate. The positive relationship I observed suggested all treatments are equal relative to invasion resistance.

The observed relationship between native and non-native abundances was variable (Psuedo r2 = 0.09) which may be attributed to unaccounted for variables (e.g. soil moisture, light, nutrients, etc.; Figure 2.5). Despite the variability, I still observed a significant negative relationship likely attributed to the heightened competitive advantage of established native vegetation (Corbin and D’Antonio 2004, Davies and Johnson In Press). As reported in other studies, combining canopy reductions with fire led to the greatest increase in understory biomass (Masters et al. 1993). Increases in native species abundance following this disturbance was likely the mechanism buffering invasion. Contrastingly, the herbicide addition killed existing understory vegetation that likely increased competitiveness of invading non-native species. Further, the lack of fire in the canopy reduction treatment limited the abundance of many dominant native species typically promoted by fire, which may have allowed the establishment of nonnative shrubs and vines. Thus, if non-native invasions are a concern it may be most beneficial to
manage for the highest abundance of native plants rather than for the greatest diversity of native plant species.

Stands with intermediate disturbances had the lowest abundance of non-native species, supporting that intermediate disturbance intensity leads to greater invasion resistance. In this ecosystem, coupling canopy reductions with fire mimics the natural disturbance regime (Pyne 1982, Frost 1998). Thus, native plant species may be better adapted to those disturbance regimes than nonnative plant species (Gutschick and BassiriRad 2003). Contrastingly, deviating from natural disturbance regimes could increase invasibility if native plants are poorly adapted to compete (Milchunas et al. 1989, D’Antonio et al. 1999). Thus, other ecosystems with plants that have evolved with differing disturbance regimes may respond similarly to their respective intermediate disturbance, which may not include canopy disturbance or fire.

Non-native invasions are one of the greatest threats to conservation; therefore, it is important to understand the role forest management techniques play in invasibility (Wilson 1997). Control and suppression of exotic plants costs an estimated 34 billion dollars each year (Pimental et al. 2005). However, that valuation is conservative because it does not consider the monetary loss associated with lost ecosystem services. Plant invasions are known to disrupt ecosystem services in many ways including displacing native plants and communities (Hiebert and Stubbendieck 1993, DeLong 2002, Moser et al 2009), affecting trophic structure (McCary et al 2016), disrupting nutrient cycling and availability (Ehrenfeld 2003, Weidenhamer and Callaway 2010), and negatively impacting wildlife species of concern (Flanders et al. 2006, Barnes et al. 2013).
Therefore, forest and wildlife managers should consider invasion potential prior to implementation of any management strategy.

**Conclusions**

Coupling canopy reductions with prescribed fire may be the best option for limiting invasibility in southern pine (Pinus spp.) forests. However, I also contend that matching sites with historically accurate disturbance regimes could lead to a greater invasibility resistance across ecosystems that evolved with differing disturbance regimes. Additionally, I suggest examining non-native invasion success based on species abundance rather than richness.
Table 2.1  Mean and standard error of the percent cover of non-native plants

<table>
<thead>
<tr>
<th>Intensity:</th>
<th>Treatment</th>
<th>June</th>
<th>August</th>
</tr>
</thead>
<tbody>
<tr>
<td>Low</td>
<td>Canopy Reduction</td>
<td>0.2805</td>
<td>0.3019</td>
</tr>
<tr>
<td></td>
<td>+Fire</td>
<td>0.1402</td>
<td>0.1675</td>
</tr>
<tr>
<td>High</td>
<td>+ Herbicide &amp; Fire</td>
<td>0.1836</td>
<td>0.2608</td>
</tr>
</tbody>
</table>

Rows with different letters indicate significant difference at p = 0.05
Figure 2.1  Conceptual figure detailing predicted invasion resistance

Conceptual figure showing the predicted relationship between disturbance intensity and non-native invasion resistance
Figure 2.2  Map of the study area

Map of the study area detailing treatments applied at the stand level.
Figure 2.3  Relationship between non-native and native species richness

The relationship between non-native and native species richness for A) June and B) August
Figure 2.4   Relationship between non-native and native species richness by treatment

Relationship between non-native and native species richness for A) Canopy reduction, B) Canopy reduction + fire, and C) Canopy reduction + herbicide & fire
Figure 2.5  Relationship between non-native and native abundance

Relationship between non-native and native species abundance during the August sampling period
Literature Cited


Elton, C. S. 1958. The ecology of invasions by animals and plants. Methuen, London, UK


CHAPTER III
COVER IS THE HABITAT COMPONENT INFLUENCING FINE-SCALE HABITAT USE OF WHITE-TAILED DEER (*ODOCOILEUS VIRGINIANUS*) AND WILD TURKEY (*MELEAGRIS GALLOPAVO*)

Introduction

Habitat quality is a foundational topic in wildlife ecology and management (Van Horne 1983). The relative importance of food availability and cover are often discussed in studies of habitat quality, leading many to focus on the impacts of improving either food availability/quality or cover on animal use. Whether food or cover is more important for influencing animal use is debatable and is likely related to which factor is limiting individual fitness. Some studies report food availability as the primary habitat component affecting animal use (Clary and Larson 1971, Coloumbe et al. 2011, Martin et al. 2012), while others report cover as more important (Hamerstrom et al. 1957, Abramsky et al. 1996, Creel et al. 2005, Lashley et al. 2015). Typical habitat quality studies are limited by time and labor leading to broad-scale data collection. This commonly leads to averaged vegetative characteristics for a treatment, assuming homogeneity throughout (Mysterud and Østbye 1999). This is especially problematic as heterogeneity may be one of the most important factors driving habitat use (Coulombe et al. 2011, Lashley et al. 2015). To accurately estimate habitat quality, animal use and vegetative characteristics should be monitored at a fine scale.
Camera trapping is a popular tool for monitoring animal use and selection, and it has been used since the early 1900s (Chapman 1927) to collect animal data in the field. Technology has improved drastically in the last 30 years, making camera trapping a more effective and economical tool for ecology studies (Rowcliffe and Carbone 2008, McCallum 2013). Camera traps are less invasive than many other techniques (Cutler and Swann 1999), allowing data to be collected without influencing animal use. Due to their increasingly low cost, researchers are now able to increase sampling intensity and apply more cameras at a finer scale than before. This provides the opportunity to evaluate fine-scale animal use.

Food and cover resources can be manipulated through forest management activities. Implementing canopy reductions allows light to reach the forest floor thereby increasing understory forage and structure (Hurst et al. 1980, Greenberg et al. 2011). However, without periodic disturbances, woody encroachment leads to degraded structure and forage availability (Crawford 1971, Harper 2007, Jackson et al. 2007). Coupling canopy reductions with periodic prescribed fire controls the woody encroachment, creating a two-tiered forest structure comprised of grasses, forbs, and woody plants (Pack et al. 1988, Masters et al. 1993, Lashley et al. 2011, McCord et al. 2014). Another common management technique is combining canopy reductions and thinning with selective herbicide application. The herbicide application removes the hardwood midstory and increases herbaceous vegetation (Godfrey and Normal 1999, Edwards et al. 2004). By using forest management techniques alone and in conjunction with one another, a wide range of vegetation conditions can be created. By coupling fine-
scale vegetation data from multiple treatments with fine-scale animal use, I can determine which vegetative characteristics animals perceive as high quality.

To address a gap in the literature and determine which vegetation characteristics drive animal use, I implemented various forest management practices in a southern pine (Pinus spp.) ecosystem to create a wide disparity in vegetative conditions. I then measured fine-scale vegetation characteristics and fine-scale animal use to determine which vegetation characteristics are perceived to be components of high quality habitat for white-tailed deer (Odocoileus virginianus; hereafter, deer) and wild turkey (Meleagris gallopavo; hereafter, turkey). I used these species due to their prevalence on the site, economic and recreational importance, and most importantly because of contradictory information regarding which conditions these animals perceive as high quality. I predicted that white-tailed deer’s perception of high quality habitat would include those areas comprised of the highest abundance of forage and cover. Additionally, I expected to see the same trend with wild turkey, wherein they would use areas with a well-developed understory comprised of high levels of herbaceous vegetation.

**Materials and Methods**

**Study site**

My study was conducted at Andrews Forestry and Wildlife Laboratory in Oktibbeha County, Mississippi. It was a 200-ha, 27-year-old loblolly pine (Pinus taeda) forest (Figure 3.1). The study area was within the flatwoods topographic region with gently sloping terrain at an elevation ranging from 90 to 101 m. The dominant soil type present on the study area was the Falkner silt loam with 0-5% slope and the average annual rainfall is ~140 cm (U.S. Climate Data).
Combinations of canopy reduction, herbicide and fire were implemented within stands to intentionally cause a wide-range of vegetation compositional and structural disparity. In 2014, 18 stands ranging from 5-22 ha in size were thinned to target basal areas (BA) of 9, 14, or 18 m²/ha producing a gradient in basal area ranging from 5-30 m²/ha at the plot level within those stands. In three stands, imazapyr and metsulfuron-methyl were applied to remove understory and midstory vegetation in October 2014. In May 2015, prescribed fire was implemented all three stands receiving herbicide and three stands receiving only thinning.

**Field Sampling**

The study area was systematically divided into a 200-m sampling grid with permanent sampling plots occurring at intersections of the grid lines (n=81). Each sampling location was approximately 169 meters apart. In June and August of 2016, I measured understory composition, visual obstruction, midstory density and composition, basal area, soft mast production and animal use (Figure 3.1). I measured understory composition and height (cm) using three 30-m point-intercept transects. Visual obstruction was measured in each cardinal direction using a vegetation profile board as described by Nudds (1977). I measured midstory density and composition using a 1/25th hectare fixed-plot, wherein I identified all midstory trees to species and measured their diameter (DBH). Basal area was measured using a 10-basal area factor (BAF) prism. Soft mast was sampled monthly using the Fruit Count method described in Lashley et al. (2014). A camera trap was deployed at each sampling location to monitor animal use (Figure 3.1) continuously during May 15, 2016 through September 15, 2016 with a one minute sampling delay between detections. I intended to obtain at least 100 detections for
robust use estimates as suggested by Lashley et al. (2018) to provide robust estimates of activity.

**Data Analysis**

To account for varying detection probabilities occurring at different sampling locations, I implemented an opposing camera design. I used a stratified random sample to select opposing camera locations (n=20) where I randomly selected 5 locations from each of the 4 subdivisions of visual obstruction observed in my data. I defined a detection (1) as an event when both cameras captured the same individual. A non-detection (0) was defined as a detection by only one of the cameras. I used the detection data to calculate detection probabilities for each camera location using a binomial generalized linear model (GLM) with animal size, midstory density, % cover grass, % cover forbs, % cover woody plants, % cover brambles as predictor variables (Version 3.3.1, R Development Core Team, 2016). I then re-sampled 10,000 possible models using the vector of estimated coefficients and the variance-covariance matrix of the model and calculated the detection probability from each of the 10,000 models for each sampling location. I used this output to create the corrected use value by dividing the raw use values (observed use at each location) by the species and location specific detection probability, resulting in 10,000 corrected use values for each location and species to accommodate error in the detection model. To model the effect of vegetation characteristics on animal use, I used a generalized additive model for location scale and shape (GAMLSS) with a zero-adjusted gamma distribution (ZAGA; Stasinopoulos and Rigby 2007). The ZAGA is a mixture distribution describing continuous positive data (a gamma distribution) while allowing that data points may have a value of 0 (a binomial distribution). I fit 10,000 corrected use
GAMLSS models, averaged their beta coefficients, and then calculated the 2.5% and 97.5% empirical quantiles to assess the effect of each variable (Table 3.2).

**Results**

Proportional coverage of functional groups I used in my model had a great deal of variability, ranging from 0.276-0.686 for grasses, 0.265-0.491 for forbs, 0.027-0.121 for woody plants, 0.093-0.894 for brambles, and 0.607-0.702 for forage plants (Table 3.1). Due to this wide range of vegetative conditions, I identified which characteristics were driving animal use.

During the 123 day sampling period, I observed 189 turkeys (0.019 observations/trap night) and 2915 deer (0.293 observations/trap night). Deer and turkey use were positively associated with percent cover of grasses, woody species, and brambles (Table 3.2), all of which constitute cover and food. Deer use was negatively associated with percent cover of preferred food plants and high levels of visual obstruction but had no relationship with percent cover of forbs and average vegetation height. Turkey use was negatively associated with fruit abundance but was not affected by visual obstruction and percent cover of forbs. Use of both species was negatively associated with percent cover of invasive species (Figure 3.2).

**Discussion**

My results indicate both deer and turkey selected for cover rather than food, which is likely a reflection of the importance of predator avoidance to prey decision rules (Goertz 1964, Cimino and Lovari 2003, Lashley et al. 2015). Predation risk represents a greater proximate fitness cost than suboptimal acquisition of resources (Brown and
Kotler 2004). Coulombe et al. (2011) demonstrated this point well with white-tailed deer when they manipulated predation risk and observed a shift in deer use from cover to forage availability. Similar decision rules related to predation risk and foraging have been demonstrated in red deer (Cervus elaphus) which selected areas with better cover (Jayacody et al. 2008). This could allow them to increase their feeding rate to overcome the more limited food resources in safe patches (Illius and Fitzgibbon 1994).

Decoupling food and cover can be difficult, particularly with generalist herbivores that can utilize plants from different growth forms for food and cover. In this case, grasses, brambles, shrubs and young trees could provide adequate cover and food resources simultaneously (Miller and Miller 2005). The negative association I observed with deer use and forage availability could be an artifact of deer using cover for forage. However, plants I chose are commonly regarded as high-quality forages often selected by deer (Miller and Miller 2005), indicating even in the case deer could consume their cover, they chose to do so at the expense of exploiting the highest availability of high-quality forages. Similarly, turkeys selected areas high in grasses, brambles, and woody species that provide high-quality brood-rearing cover (Godfrey and Norman 1999, McCord et al. 2014) but may also have some foraging values provided by seeds, soft mast, and invertebrates (Shelton and Edwards 1983, Baughman and Guynn 1993). On my study area, deer and turkey could effectively forage within cover due to cover resources simultaneously providing food. However, they were unable simultaneously exploit cover and forage in areas comprised of the highest quality food resources due to the lack of cover.
Considering life history when evaluating importance of cover to animal selection is important. My sampling period encompassed the nesting and brood-rearing seasons for turkey and fawning season for deer (Miller et al. 1998, Campbell et al. 2016.) Selection of cover is often strongest during reproduction because of the relative vulnerability of young animals to predation risk (Kunkel and Mech 1994, Miller et al. 1998, Paisley et al. 1998, Chitwood et al. 2015, Lashley et al. 2015). Because poult and neonate predation are important to population dynamics (Vangilder and Kurzejeski 1995, Roberts and Porter 1996, Rolley et al. 1998, Chitwood et al. 2015), the evolved response of strengthened animal selection of cover to maximize recruitment is to be expected. In some species, these behaviors can be evolutionarily ingrained such that animals strongly select cover perceived as high-quality, despite suffering from higher rates of predation (Chitwood et al. 2017). Similarly, my study demonstrates the strong selection of turkeys for areas with taller understory vegetation and areas with high understory woody composition, which positively influences nesting and brood-rearing success (Hubbard et al. 2001, Streich et al. 2015).

Deer and turkey selected areas with vegetative conditions that are consistently produced by combining canopy reductions and prescribed fire. Due to the economic value of deer and turkey, it is important for land managers and landowners to use management techniques that maximize habitat quality for these species. As mentioned in Haines et al. (2001), management of southeastern forests is moving towards reliance on herbicide applications partially because of the lack of recurring prescribed fire. This management trend is concerning, as herbicide applications did not provide the same high quality cover resources for either species that were produced following prescribed fire.
Future research is needed to compare the fitness consequences of herbicide replacement of prescribed fire.

**Conclusions**

This study adds to the literature base stating cover as an important component of deer and turkey habitat in many systems. Combining thinning operations and prescribed fire consistently created the most highly desired cover for both deer and turkey. Moving forward, research should focus on the fitness impacts of cover and food.
Table 3.1  Mean proportional coverage of functional groups among the treatments and the standard error (SE).

<table>
<thead>
<tr>
<th>Treatment</th>
<th>Grass (SE)</th>
<th>Forb (SE)</th>
<th>Woody (SE)</th>
<th>Bramble (SE)</th>
<th>Forage (SE)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Canopy Reduction</td>
<td>0.276 (0.023)</td>
<td>0.265 (0.026)</td>
<td>0.121 (0.009)</td>
<td>0.693 (0.039)</td>
<td>0.695 (0.055)</td>
</tr>
<tr>
<td>+ Fire</td>
<td>0.311 (0.017)</td>
<td>0.418 (0.047)</td>
<td>0.096 (0.015)</td>
<td>0.894 (0.066)</td>
<td>0.702 (0.085)</td>
</tr>
<tr>
<td>+ Herbicide &amp; Fire</td>
<td>0.686 (0.038)</td>
<td>0.491 (0.053)</td>
<td>0.027 (0.005)</td>
<td>0.093 (0.024)</td>
<td>0.607 (0.081)</td>
</tr>
</tbody>
</table>

As can be seen from the table, the treatments produced a wide disparity in vegetative conditions on the property.
Table 3.2  Beta coefficient means, lower (LCI), and upper (UCI) empirical quantiles for deer and turkey use.

<table>
<thead>
<tr>
<th>Species</th>
<th>Variable</th>
<th>Estimate</th>
<th>LCI</th>
<th>UCI</th>
<th>Association</th>
</tr>
</thead>
<tbody>
<tr>
<td>Deer</td>
<td>Intercept</td>
<td>-0.5411</td>
<td>-2.9530</td>
<td>1.0454</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Midstory Density</td>
<td>0.0038</td>
<td>0.0020</td>
<td>0.0053</td>
<td>+</td>
</tr>
<tr>
<td></td>
<td>% Grass</td>
<td>10.2692</td>
<td>5.7517</td>
<td>24.3182</td>
<td>+</td>
</tr>
<tr>
<td></td>
<td>% Woody</td>
<td>10.3206</td>
<td>4.3374</td>
<td>32.9345</td>
<td>+</td>
</tr>
<tr>
<td></td>
<td>% Brambles</td>
<td>4.0914</td>
<td>2.5357</td>
<td>7.3183</td>
<td>+</td>
</tr>
<tr>
<td></td>
<td>% Forb</td>
<td>0.1973</td>
<td>-5.4173</td>
<td>2.1888</td>
<td></td>
</tr>
<tr>
<td></td>
<td>% Forage</td>
<td>-0.8498</td>
<td>-4.2002</td>
<td>-0.2510</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Average Height</td>
<td>-0.4490</td>
<td>-5.1791</td>
<td>0.0946</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Basal Area</td>
<td>0.0183</td>
<td>0.0112</td>
<td>0.0777</td>
<td>+</td>
</tr>
<tr>
<td></td>
<td>% Invasive</td>
<td>-0.0174</td>
<td>-0.1042</td>
<td>-0.0040</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Visual Obstruction</td>
<td>-0.2100</td>
<td>-0.2720</td>
<td>-0.1269</td>
<td>-</td>
</tr>
<tr>
<td>Turkey</td>
<td>Intercept</td>
<td>-6.1952</td>
<td>-10.0034</td>
<td>-1.9289</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Midstory Density</td>
<td>0.0054</td>
<td>0.0017</td>
<td>0.0085</td>
<td>+</td>
</tr>
<tr>
<td></td>
<td>% Grass</td>
<td>12.5564</td>
<td>7.2856</td>
<td>17.4904</td>
<td>+</td>
</tr>
<tr>
<td></td>
<td>% Woody</td>
<td>6.1237</td>
<td>0.4951</td>
<td>11.4030</td>
<td>+</td>
</tr>
<tr>
<td></td>
<td>% Brambles</td>
<td>4.2570</td>
<td>1.9237</td>
<td>6.6470</td>
<td>+</td>
</tr>
<tr>
<td></td>
<td>% Forb</td>
<td>1.4495</td>
<td>-1.7802</td>
<td>4.7599</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Soft Mast</td>
<td>-0.0002</td>
<td>-0.0004</td>
<td>0.0000</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Average Height</td>
<td>2.1142</td>
<td>1.0787</td>
<td>3.6615</td>
<td>+</td>
</tr>
<tr>
<td></td>
<td>Basal Area</td>
<td>0.0126</td>
<td>0.0066</td>
<td>0.0233</td>
<td>+</td>
</tr>
<tr>
<td></td>
<td>% Invasive</td>
<td>-0.0072</td>
<td>-0.0104</td>
<td>-0.0051</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Visual Obstruction</td>
<td>0.0177</td>
<td>-0.0027</td>
<td>0.0295</td>
<td></td>
</tr>
</tbody>
</table>

Variables that are different from zero are designated with either a + (selecting for) or – (avoiding).
Included in the data collection was 3 30-m point intercept transects (a), visual obstruction readings taken in 4 cardinal directions at 10m from plot center (b), a 1/25th –ha midstory sample (c), 1 randomly placed 1m² biomass collection (d) and a 25-m fruit abundance transect (a). Additionally, a camera trap was placed at each location to monitor animal use (e).
Figure 3.2  Mean beta coefficients for variables influencing deer (A) and turkey (B) habitat use.

Values above baseline 0 indicate positive influence on use and values below indicate negative influence. Error bars represent 2.5% and 97.5% empirical quantiles.
Literature Cited


CHAPTER IV
CONCLUSIONS AND SYNTHESIS

Conservation of ecosystems and species requires active forest management. Using techniques that most closely mimic natural disturbance regimes may best resist non-native invasion. In my study system, non-native plant abundance was the lowest in stands that were treated with canopy reductions and prescribed fire. Invasion resistance was driven by the abundance of native plants rather than species richness, contrary to leading hypotheses. Future research should focus on the impacts of other common management practices on non-native plant invasions.

Coupling prescribed fire and thinning operations also created the most desirable conditions for wild turkey (*Meleagris gallopavo*) and white-tailed deer (*Odocoileus virginianus*). Deer and turkey selected areas comprised of high levels of cover that simultaneously provided food resources. This allowed animals to forage efficiently while avoiding predation. No other treatment applied on my study area consistently created the desired composition and structure. I recommend that managers and landowners focus on management of cover resources such as brambles, native grasses, and understory woody plants to simultaneously provide cover and food resources. Future research should focus on independent and combined fitness impacts of cover and food.