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Effects of Large Scale Growing Season Prescribed Burns on Movement, Habitat Use, Productivity, and Survival of Female Wild Turkey on the White Rock Ecosystem Restoration Project of the Ozark-St. Francis National Forest

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Abstract

Restoration of woodland and savanna ecosystems has become a common management strategy in the Central Hardwoods region. Over the past two decades forest managers have implemented woodland and savanna restoration at the landscape level (≥10,000 ha), especially using early growing season prescribed fire. The implementation of the restoration strategy has coincided with declines of Eastern Wild Turkey (*Meleagris gallopavo silvestris*) in many treated areas causing concern that early growing season prescribed fire was impacting wild turkey. We initiated our study to examine the effect of woodland and savanna restoration on the ecology and habitat of wild turkey in the Ozark Highlands. We used 67 female wild turkey fitted with 110 g Global Positioning System (GPS) Platform Transmitting Terminals between 2012 and 2013 to document nest-site selection and survival, estimate annual and seasonal home ranges, examine pre-incubation habitat use, and assess the impacts of management practices on forest structure. Nest-sites had higher visual concealment, higher slope, and more woody ground cover than non-nest-sites. We also found that wild turkey nest survival increased as the amount of visual concealment increased and survival decreased as the distance from a road increased. We documented wild turkey home ranges that were among the largest reported for the species and were larger than those documented before woodland and savanna restoration. We found wild turkey selected habitat during the pre-incubation period that was more diverse in canopy cover, in a transitional state and in small patches. Wild turkey subsequently selected habitat for nest-sites that had similar characteristics but were in larger patches. We also found that landscape level early growing season prescribed fire had not created woodland or savanna conditions across the landscape and likely would require more time (≥25 years). In conclusion wild turkey populations have not benefited from current woodland and savanna restoration. However, if restoration were having the desired outcome the impact on wild turkey population may be

different. We provide a description of all variables used (Appendix I), morphometric and handling data for all captured wild turkey (Appendix II), data sets for nest-site selection (Appendix III), nest survival (Appendix IV), pre-incubation habitat selection (Appendix V), and vegetation data collected throughout the study area from 2011 to 2013 (Appendix VI).

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Supplementary Material

Preface

This dissertation was written in journal style and format and organized into three chapters intended for independent publication. Style and format may vary between chapters and some text may be repeated. These chapters are preceded by an introduction and conclusion chapter.

Introduction

The Eastern Wild Turkey (*Meleagris gallopavo silvestris*) is one of the most sought after and economically important gallinaceous birds in eastern North America (Dickson 2001). The Eastern Wild Turkey is one of five recognized sub-species in North America, the Rio Grande (*M. g. intermedia*), the Mirriam's (*M. g. meriami*), the Florida (*M. g. Osceola*), and the Gould's (*M. g. Mexicana*). Populations of all sub-species were almost extirpated from the landscape after European settlement of North America. Extirpation was likely a result of changes in land use, deforestation and increased mortality as a result of market hunting causing wild turkey to reach their lowest number in the 1930's (Mosby 1975). However wild turkey benefited from the great depression in the 1930's when former agriculture land began to revert to early successional habitat. In the 1930's and 1940's wild turkey also benefited from new conservation efforts and legislation, and after the Second World War many state and federal wildlife agencies began the process of restoring wild turkey to the former range.

Initial restoration efforts consisted of releasing pen-raised wild turkey into areas without wild turkey populations. It was eventually realized that the release of pen-raised wild turkey was much less successful than the release of wild-captured birds from other stock populations (Kennamer et al. 1992). Wild turkey benefited most from the invention of the rocket-net in the 1950's (Kennamer et al. 1992). The rocket-net allowed wildlife agencies to capture and relocate wild-raise wild turkey more efficiently and establish new populations with greater success. After the invention of the rocket net, wildlife agencies began trap and transplant operations throughout the eastern wild turkey's range including Arkansas. Arkansas was one of the first states to begin wild turkey conservation efforts by passing protective legislation as early as 1918 (Widner 2007). Arkansas also initiated reintroduction projects as early as 1932 but made significant restoration efforts from 1960 to 1990 increasing the states wild turkey population from approximately 7,000 wild turkey to over 100,000 by the end of the 20th century (Widner 2007).

After significant population expansion in Arkansas through the second half of the 20th century wild turkey populations as indicated by harvest have declined over the past 10 years. Wild turkey harvest decreased from their highest statewide totals, 19,647 in 2003, to 10,111 wild turkey harvested in 2010 at the initiation of this study. During this period wild turkey harvest declined across all physiographic regions of Arkansas but was most dramatic in the Ozark Highlands (Figure 1). It was realized most on White Rock wildlife management area located in the Boston Mountains portion of the Ozark Highlands and the focus of this study (Figure 2).

Across the Ozark Highlands, especially the Boston Mountains, over the past 10 to 15 years forest managers have initiated landscape level woodland and savanna restoration. Restoration efforts aim to revert closed canopy upland forest to historical woodland and savanna conditions mostly through the implementation of early growing season prescribed fire. Woodlands and savannas were once prevalent throughout the Central Hardwoods region including the Ozark Highlands and the Boston Mountains (Nuzzo 1985). They represented a transition zone between the tall grass prairie in the Central Plains and the closed canopy deciduous forests of eastern North America (Dyksterhuis 1957; Anderson 1983; Nuzzo 1985). Woodland and savanna ecosystems consisted of open canopies and diverse understories of grasses, forbs, and some woody shrubs, and were maintained by disturbance, typically fire. (McPherson 1997; Taft 1997; Anderson et al. 1999). After the fire suppression movement and land use changes of the 19th and 20th centuries woodlands and savanna ecosystems were reduced to approximately 1% of their original extent (Nuzzo 1985).

In the Ozark Highlands the loss of woodland and savanna ecosystems and the fire regimes that maintained them coincided with increased populations of European settlers (Cutter and Guyette 1994). Foti (2004) approximated that 47% of the Boston Mountains in the Ozark Highlands region were once woodland ecosystems and another 15% were in herbaceous prairie. These ecosystems supported a variety of diverse plant and wildlife species that were dependent on the transitional nature of these ecosystems. Until recently, only a small amount of these ecosystems were still present on the landscape and the fire regimes that maintained them were totally absent.

Restoration of woodland and savanna ecosystems was initiated approximately 10 to 15 years ago throughout the Ozark Highlands by federal and state management agencies. In the Boston Mountains the United States Department of Agriculture, Forest Service selected an area, the White Rock Ecosystem Restoration Area (WRERA), to implement woodland and savanna restoration in 2002. Wild turkey harvest began to decline shortly after the implementation of restoration through landscape scale early growing season prescribed fire (Figure 2). As restoration has progressed up to the initiation of this study so has the decline in wild turkey harvest leading to concern that landscape scale early growing season prescribed fire were negatively affecting wild turkey populations.

Wild turkey have previously been studied in Arkansas and on the WRERA. Badyaev (1994) examined the ecology of female wild turkey on the WRERA before its establishment and before woodland and savanna restoration. This study documents habitat requirements, productivity, survival rates, and home ranges and contributed significantly to our understanding of eastern wild turkey ecology in Arkansas and throughout its range. Moore (1995) and Thogmartin (1998) both studied eastern wild turkey in the Ouachita Mountains of Arkansas.

Moore (1995) focused on the nest-site selection process of eastern wild turkey in the Ouachita Mountains. Thogmartin (1998) examined space use, survival, and habitat selection of female eastern wild turkey. All of these previous research efforts have provided essential background information used in our study.

We initiated our study on the WRERA in 2011 basing much of our study design and many of our hypotheses on the previous efforts of Badyaev (1994), Moore (1995), and Thogmartin (1998) while developing new designs and hypotheses to address the current aspects of woodland and savanna restoration. Our study was intended to address five main objectives: 1.) examine habitat selection, especially nest-site selection, of female wild turkeys at multiple spatial scales with respect to landscape scale early growing season prescribed fire, 2.) document pre-nesting movements of female wild turkey and relate those movements to productivity, 3.) estimate period and annual female wild turkey survival and productivity, 4.) compare research results of Badyaev (1994) to those from the previous study on later established WRERA, and 5.) develop management recommendations to enhance nesting habitat availability, female survival, and recruitment in the Central Hardwoods Region. This dissertation was intended to serve as the final report of research findings for the study and was written with the intent of publishing each chapter separately. Chapter 1 was written in the journal format of *Forest Ecology and Management* for submission with David G. Krementz as coauthor. Chapter 2 was written in the journal format of *Landscape Ecology* for submission with David G. Krementz as coauthor. Chapter 3 was written in the journal format of *Ecological Applications* for submission with David G. Krementz as coauthor.

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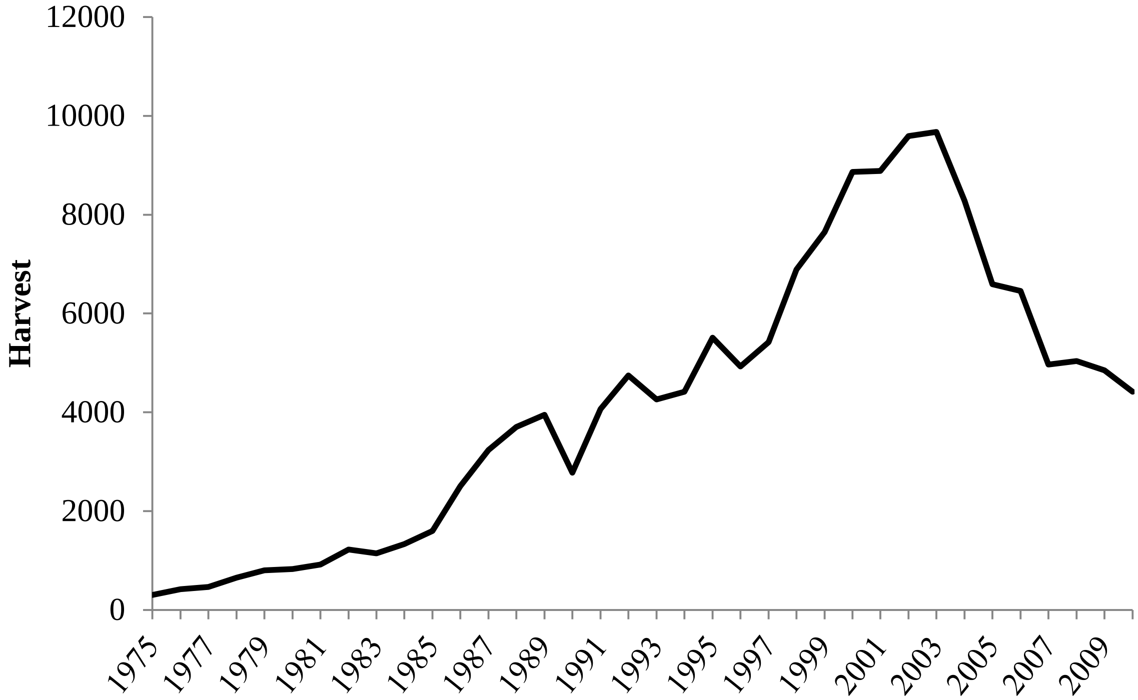

Figure 1. Eastern Wild Turkey harvest in the Ozark Highlands physiographic region by year from 1975 to 2010.

Figure 2. Eastern Wild Turkey harvest on White Rock wildlife management area in the Ozark Highlands physiographic region by year from 1975 to 2010.

Chapter I

Impacts of upland hardwood ecosystem restoration through early growing season prescribed fire on Eastern Wild Turkey Nest Ecology in the Ozark Highlands, Arkansas, USA

Henry Tyler Pittman

And

Dr. David G. Krementz

Abstract

Landscape level woodland and savanna restoration is becoming an increasingly common management strategy in the Central Hardwoods region and has been implemented on the White Rock Ecosystem Restoration Area (WRERA) of the Ozark-St. Francis National Forest in Arkansas since 2002. Management has primarily consisted of landscape scale (500-2,500 ha) early growing season prescribed fire and has coincided with declines in eastern wild turkey (*Meleagris gallopavo silvestris*). We initiated this study to assess how management has impacted the quality and/or quantity of wild turkey nesting habitat on the WRERA. We marked 67 wild turkey hens with 110 g Global Positioning System Platform Transmitting Terminals in 2012 and 2013 on the WRERA to examine nest-site selection and estimate nest survival. We used mixed-effect logistic regression to determine what habitat characteristics hens selected. We found that hens selected initial nest-sites with higher visual concealment (0-1 m in height), percent slope, and woody ground cover. We also found that as time since prescribed fire increased so did visual concealment (0-1 m) at initial nest-sites. Renest-sites were best characterized by higher visual concealment (0-1 m) and fewer small shrubs. Our best approximating nest survival model indicated that as visual concealment (0-1 m) increased so did nest survival, but as distance from road increased nest survival decreased. We found hen success for adult wild turkey on WRERA was 23.5%, among the lowest reported. Overall, nest-sites had some woodland and savanna characteristics but low nest survival and hen success suggested that the quality of nesting habitat at WRERA may still be a limiting population factor.

1. Introduction

Prescribed fire has only recently been returned to upland hardwood ecosystems after the fire suppression movement of the twentieth century. During fire suppression many upland hardwood ecosystems transitioned from open woodlands and savannas to closed canopy forests (Batek et al. 1999; Dey et al. 2005). Over the past decade efforts to reintroduce fire and restore historical woodland and savanna conditions to upland hardwood ecosystems have increased, especially in the Ozark Highlands. This practice has been termed “restoration,” as its intent is to restore historical conditions from when fire was present and frequent on the landscape (Greenberg et al. 2013). “Restoration” in the Ozark Highlands typically consists of the use of early growing season prescribed fire over large spatial extents to create heterogeneity in understory and canopy vegetation. Early growing season prescribed fire occurs usually in March and April in the Ozark Highlands of Arkansas. The effects of prescribed fire on vegetation responses have been documented in several upland hardwood ecosystems, but more information on severity and vegetation responses over large extents and varying topography is needed to fully understand the effects of prescribed fire on wildlife.

Prescribed fire can be a useful tool for the recruitment and retention of early successional bird species in eastern upland hardwood ecosystems (Lanham et al. 2006; Greenberg et al. 2007; Greenberg et al. 2013). However, these findings have often been in conjunction with mechanical habitat management and do not address the current conditions of landscape scale restoration. Since the implementation of prescribed fire and restoration bird responses are much less understood in the Ozarks and Central Hardwood region (Thompson et al. 2012). Reidy et al. (2014) found that woodland generalist and early successional specialist have benefited from restoration prescribed fire in the Missouri Ozarks while ground nesting species such as worm eating warbler (*Helmitheros vermivorus*) have higher densities in forest sites with less influence from restoration efforts and prescribed fire (Reidy et al. 2014). These studies have also noted potential risks of prescribed fire to ground-nesting species like destruction of nests and alterations of important habitat structure. Current research also does not examine the effect of restoration and prescribed fire on habitat characteristics critical to nesting of ground nesting species in the Ozarks and Central Hardwoods region.

Reproductive performance is an important factor in the maintenance of wildlife populations especially in ground-nesting birds like the wild turkey (Martin 1993; Thogmartin 1998). Nesting habitat selection and subsequent nest survival are critical aspects of the reproductive performance of Eastern wild turkeys (*Meleagris gallopavo silvestris*). These aspects of wild turkey life history have been well documented with respect to prescribed fire and other management practices, but predominately in Coastal Plain pine (*Pinus sp*) or mixed pine/hardwood ecosystems (Day et al. 1991; Moore et al. 2010; Williams 2012). Prescribed fire has been studied in South Carolina Coastal Plain pine ecosystems and been found to increase potential nesting habitat for wild turkeys (Moore et al. 2010). More specifically early growing season prescribed fire has been studied in Georgia Coastal Plain pine ecosystems and found to have minimal negative impacts since wild turkey had high renesting rates and nest survival (Williams 2012). However few other studies have examined nest-site selection and nest survival in upland hardwood ecosystems of the Central Hardwoods region (Vangilder et al. 1987; Badyeav 1994; Vangilder et al. 1995). Most importantly, to our knowledge, nesting habitat selection and nest survival have not been examined in relation to woodland and savanna restoration currently being implemented in the Central Hardwoods region.

1.1 Objectives

Our research stemmed from this lack of knowledge about the overall response of ground nesting birds to early growing season prescribed fire used for woodland and savanna restoration in the Central Hardwoods region. Also, declining productivity and harvest of wild turkey in the Ozark Highlands that coincided with the implementation of woodland and savanna restoration has motivated efforts to better understand the role of woodland and savanna restoration in the nest habitat selection process and subsequent nest survival of ground nesting birds like the wild turkey. Since the implementation of woodland and savanna restoration, wild turkey harvest decreased 60% in the Ozark highlands and 84% in our study area (K. Lynch, Arkansas Game and Fish Commission, unpubl. data). Our research consists of two objectives: 1) examine nest-site selection of wild turkey at multiple spatial scales and its relationship to prescribed fire and 2) examine nest survival and its relationship to habitat and/or management practices.

1.2 Study area

The WRERA consists of 16,380 ha of upland hardwood and pine ecosystems in northwest Arkansas, United States of America (USA; Fig 1). It is part of the larger main division of the Boston Mountain Ranger District (41,400 ha) on the Ozark-St. Francis National Forest. The WRERA is a high priority woodland and savanna restoration area for the United States Department of Agriculture (USDA) Forest Service. Historically the WRERA was dominated by woodlands and savannas maintained by frequent disturbances by fire (Chapman et al. 2006; Guyette et al. 2006; Foti 2004). After fire suppression during the 20th century, the WRERA became dominated by closed canopy hardwood forests of various oak (*Quercus sp.*) and hickory (*Carya sp*.) species. Understories consisted of canopy species regeneration, black gum (*Nyssa sylvatica*), flowering dogwood (*Cornus florida*), Carolina buckthorne (*Rhamnus caroliniana*), blackberry (*Rubus spp.*) and devil's walking stick (*Aralia spinosa*) (USDA, NRCS 2014). Pine ecosystem canopies were dominated by short leaf pine (*Pinus echinata*) while understories consisted of hardwood and pine regeneration. Since 2002, the WRERA has been intensively managed with large scale early growing season prescribed fire to restore historic woodland and savanna conditions. Management prescription for WRERA describes woodlands as having "open canopies, sparse mid-stories and well-developed understories typically dominated by grasses and forbs, but also may become shrubby between fires and have a significant woody component" with 40-60% canopy closure (ONF Forest Plan). The prescription also describes the management techniques to achieve these objectives including mechanical canopy removal, herbicide, and fire treatments. However, mechanical and herbicide treatments have been implemented on ≤ 5% of the restoration area since 2002. The WRERA is divided into 16 prescribed burn units ranging from 467 to 1,670 ha in size. Early growing season prescribed fire has been implemented on a 3 to 5 year rotation in 15 of the 16 prescribed burn units, and all units have received 1 to 4 prescribe fire treatments since 2002.

2. Methods

2.1 Nest monitoring and habitat measurement

We trapped wild turkey in and around the WRERA using rocket nets in late winter from 2011-2013. We marked all captured wild turkeys with an aluminum leg band and hens were fitted with 110 gram Platform Transmitting Terminal (PTT) with Global Positioning System (GPS) capabilities and Very High Frequency (VHF) radio transmitter (North Star Science and Technology LLC, King George, Virginia) attached using a back-pack-type harness (Kurzejeski et al. 1987). The PTTs recorded GPS locations four times per day during the 2011 and 2012 nesting seasons and eight times per day during the 2013 nesting season. We estimated GPS location accuracy to approximately 15 m. Our PTTs transmitted GPS locations via the ARGOS satellite system (CLS America) every 120 hrs during 2011 and 2012 and every 48 hrs during 2013. We determined nest-sites when >3 consecutive GPS locations were <20 m of each other during the nesting season (April-July). We approximately located nest-sites to within 50 m on the ground using the VHF transmitters. We monitored nests using daily GPS and VHF locations until the nest was abandoned or hatched. Then we recorded vegetation measurements within 48 hrs at nest-sites that were definitively located.

We recorded 8 ground collected habitat variables and 9 habitat variables derived from a Geographic Information System (GIS). We collected vegetation measurements using a 10 m radius circular plot centered at bird locations before nest initiation, random locations 50-100 m from pre-nest initiation bird locations, nest-sites, 40 m from nest-sites in a random direction, and 300 m from nest-sites in a random direction (Badyeav 1994). We recorded visual concealment in classes from 0 to 2 m in height (Nudds 1977), percent ground cover type (Daubenmire 1959), counts of live stems in diameter at breast height (dbh) classes, understory height (m), canopy cover (Lemmon 1956), slope (%), slope position (% from bottom), and aspect. Counts of live trees were made for size classes 0-15 cm, 16- 30 cm and >30 cm dbh and shrubs were counted for size classes ≤5 cm and >5 cm ground diameter. Our GIS based habitat variables consisted of time since prescribed fire (yrs), elevation (m), distance (m) to nearest road, distance (m) to nearest stream, and mechanical forest treatment, nearest treatment type, USDA Forest Service cover type, stand age (yrs), and site index.

2.2 Nest-site selection

We modelled nest-site selection at micro-habitat (<10 m) and management scales (i.e. stand, burn unit, etc.) with each scale being represented by different habitat variables. Models were analyzed with generalized linear mixed-effects models from the binomial family (function "glmer", family = binomial, link = "logit"; R Core Development Team, 2010). All models were of the same form with a binomial response (nest-site vs. non-nest-site, n=49 and n=521) and some suite of habitat predictor variables. Initial nest attempts (n=49) and renest attempts (n=16) were modeled separately. We also modeled selection at distances of 40 m and 300 m from nest-sites to examine selection patch size. We developed a micro-habitat candidate model set (n=17) based on previous findings of Badyaev et al. (1996), Thogmartin (1999), and Byrne and Chamberlain (2013). We also developed a management scale candidate model set (n=9) based on previous findings of Badyaev et al. (1996), Thogmartin (1999), and Byrne and Chamberlain (2013) Micro-habitat models were developed to understand the type and quantity of nesting habitat (Martin 1993). Management scale models were developed to determine if management practices were most influential in developing quality nesting habitat, indicated by the selection of certain management treatments as nesting habitat. The full candidate model sets are available from the authors upon request. A random effect for individual was included in each model to account for unequal sampling and variability amongst individuals (Gillies et al. 2006). We evaluated models using Akaike's Information Criterion (AIC_c) corrected for finite sample sizes (Burnham and Anderson 2002). Model fit was assessed using the Lemeshow-Hosmer goodness-of-fit test statistic (Lemeshow and Hosmer 1982). Models with $\Delta AIC_c \leq 4$ were model averaged and all non-zero parameters were reported. Non-zero parameter estimates are those where the 95% confidence intervals (CI) do not overlap zero. We used model averaging to account for any model selection uncertainty among top models.

2.3 Nest survival

Nests were monitored on a daily basis using a combination of GPS and VHF locations of incubating hens. We estimated nest daily survival rates (DSR) and nest success rate (NSR) in Program MARK (Dinsmore et al. 2002; Rotella et al. 2004) using both ground collected and GIS derived habitat variables. DSR was the probability that a nest would survive from one day to the next. NSR was the probability that a nest would survive the full 28 day incubation period. We developed a candidate model set (n=16) based on the results of previous research by Badyaev et al. (1996), Thogmartin (1999), and Byrne and Chamberlain (2013). Our full candidate model set included models based on physiological condition, behavioral status, individual characteristics, and habitat characteristics. Physiological models included variables describing age, behavior, and physiological condition represented by winter mass (kg). Age was defined as adults, individuals having experienced 2 or more breeding seasons, and juveniles, individuals in their first breeding season. Models including behavioral variables tested the hypothesis that greater dispersal distances from the winter range by females correlated with higher nest success (Badyaev 1994). Habitat based models included variables for percent visual concealment, ground cover, stem counts, distance to roads, time since prescribed fire and management types. Models were compared using AIC_c (Burnham and Anderson 2002; Appendix B).

3. Results

3.1 Nest-site selection

Initial nest attempts were best differentiated from non-use-sites by two top models all of which included percent visual concealment (0-1 m), distance to roads, time since fire, percent woody ground cover, and an interaction term for percent visual concealment (0-1 m) and time since fire (Table 1; sum of model weights $\sum w_i$=0.78). Based on model averaged parameter estimates, we found that nest-sites were positively associated with percent visual concealment 0 to 1 m above ground level (β=4.24, SE=1.00), percent woody ground cover (β =1.36, SE=0.58), and percent slope (β =1.85, SE=0.62; Table 2; Fig 2). We found that as time since fire increased (β =3.77, SE=1.78) the percent visual concealment (0-1 m) increased in a curvilinear fashion until it approached 100% (Fig 3). When we examined nest patch size, we found nest-sites had higher visual concealment (0-1 m) than at locations sampled at 40 and 300 m from the nest-sites indicating small nest patches of higher visual concealment (0-1 m).

Renest attempts were best differentiated from non-use-sites by six top models all of which included percent visual concealment (0-1 m) and distance to roads (Table 3). Other variables included in top models were time since fire, counts of small shrubs and medium trees and an interaction term of counts of small shrubs and time since fire. Model averaging resulted in only two non-zero parameter estimates, visual concealment (0-1 m) (β=2.12, SE=0.96) and count of small shrubs (β=-3.48, SE=1.57) which suggested that renesting hens were locating nests at sites with more visual concealment (Table 2). In initial nest attempt and renest attempt management based models did not perform better than the intercept only models indicating those models did not differentiate between nest-sites and available sites.

3.2 Nest survival

We identified one best approximating model of nest survival that included a temporal trend term and variables for percent visual concealment (0-1 m) and distance from road (m) (Table 4). We removed age models from the model set after GLM results confirmed there was no difference in nest habitat between adults and juveniles. We also removed age models because there were no successful nests and a small sample size in the juvenile age class that caused models to over fit the data based on the difference between age classes. In our top model all parameter estimates were significantly different than zero. The temporal trend (β = -0.07, 95% CI [-0.12,-0.03]) indicated that nest survival decreased over the 28 day incubation period. Parameter estimates also indicated that nest survival increased (β = 1.78, 95% CI [0.38, 3.19]) with increased visual concealment (0-1 m) and nest survival decreased with increased distance from a road (β = -0.005, 95% CI [-0.009, -0.0009]). However, the parameter estimate for distance to roads suggested very little influence on nest survival. Our top model estimated nest success as 0.19 (95% CI [0.11, 0.31]) and daily nest survival ranged from 0.98 (95% CI [0.96, 0.99]) on day 1 of incubation to 0.88 (95% CI [0.80, 0.93]) on day 28 of incubation.

4. Discussion

We found that visual concealment, a micro-habitat characteristic, was the most critical aspect of nest-site selection by hens in our study area. We also found that in years after a prescribed fire event, the degree of visual concealment increased and plateaued several years after a fire event. Since hens selected nest-sites with more visual concealment than non-use-sites, this relationship could play an important role in optimizing fire regimes to benefit wild turkey and other ground nesting birds. We determined for the hens that renested (22.5%) visual concealment was an important variable in selecting a nest-site. We also found hens selected initial nest-sites with higher levels of woody stemmed vegetation. Woody stemmed vegetation was comprised of woody vines and canopy seedlings approximately 1-2 year old. Selection of this type of woody stemmed vegetation could be a result of hens selecting a nest-site before green up in the spring. Persistence of woody vine structure and young seedlings from the previous growing season into the spring may be an indicator hens use in selecting a nest-site early in the reproductive period. Renesting hens selected nest-sites with lower counts of small shrubs that were <4 cm in diameter at ground level, and typically consisted of 2-4 year old saplings and other woody free standing vegetation. Lower counts of small shrubs at renest-sites suggest hens were able to satisfy their need for high visual concealment with other vegetation types that may be more available later in the season. Renest-site selection occurs during the growing season and provides hens the opportunity to select a site based on current conditions as opposed to the likely future condition of initial nest-sites. We also found hens selected initial nest-sites that had higher percent slope which could be a factor in the behavior and intensity of prescribed fire implementations (Rothermel 1983).

Our analysis of patch size indicated that hens were selecting nest-sites based on visual concealment patches less than 80 m in diameter or on transition zones between different habitat patches. These findings suggest that habitat heterogeneity is a potentially important component of ideal nesting habitat and selection of that habitat. Hens could potentially find these patches as a result of prescribed fire, tree fall, insect damage, and/or other forest disturbances. In our study area large scale prescribed fires over differing topography could potentially produce this patch heterogeneity. However, more information is needed on vegetation responses in the study area to determine if this is the case. Also, most studies of prescribed fire impacts on wildlife are at small scales and/or experimental in design, subsequently little research on prescribed fires at the landscape scale (≥500 ha) exists to draw information about patch heterogeneity (Greenberg et al. 2007, 2013; Thompson et al. 2012; Reidy et al. 2014). Our study is the only study we are aware of that has examined prescribed fire impacts on wildlife and vegetation dynamics at such a large scale, and our findings are more likely to represent wild turkey responses to prescribed fire under current management prescriptions and scales for woodland and savanna restoration.

Wild turkey nesting ecology has previously been examined in Arkansas three times including once previously on the WRERA. All three of these studies were conducted before the reintroduction of early season prescribed fire in those systems. In the early 1990's WRERA was managed under even-aged timber management strategy and prescribed fire was absent from the landscape. At that time and under those management conditions hens selected nest-sites with high visual concealment (0-1 m), and greater counts of small shrubs and medium trees (Badyeav 1994). We observed the same selection of nest-sites with high visual concealment (0-1 m), but did not observe the selection of sites based on small shrubs and medium trees. The absence of small shrubs and medium trees from our top models could be a result of the reintroduction of prescribed fire changing the availability of these characteristics on the landscape to provide visual concealment (0-1 m). Small shrubs and medium trees may have been important under fire suppression when hens likely relied on the early succession habitat created by mechanical timber harvest and the increased light conditions created by road sides to satisfy the requirement of high visual concealment for a nest-site. Badyaev (1994) also observed hens selected nest-sites that were closer to roads, had greater over-story density, and were in habitat patches approximately 80 m in diameter. We did not find that hens were selecting nest-sites that were closer to roads or had greater over-story density than available sites. However, we did observe hens selecting similar sized patches of nesting habitat suggesting patches of approximately 80 m in diameter are optimal nest patches or the availability of different size nest patches has not changes since the implementation of early growing season prescribed fire.

Working in the nearby Ouachita Mountains of Arkansas, Moore (1995) and Thogmartin (1999) found that hen nest-sites were located close to roads and had strong selection for short-leaf pine (*Pinus echinata*). We found that neither distance to roads nor cover type were important in the selection of nest-sites. However, the lack of selection by cover type and no apparent selection of short-leaf pine on the WRERA was possibly because the community composition in

Ozark Highlands differs from the community composition in the Ouachita Mountains. In the Ouachita Mountains, hens were also observed to select patches of nesting habitat much larger than those observed in the Ozark Highlands (Thogmartin 1999). This patch size difference could be a difference related to management practices or ecosystem characteristics. Forest management in the Ouachita Mountains is focused toward forest products in pine ecosystems. This consists of more mechanical disturbance with a small interval (~ 10 years) between disturbances. Mechanical disturbance tends to be applied at the stand scale and are assumed uniform across the area of application.

Numerous other studies have examined hen nest-site selection in a wide variety of ecosystems and under a various management regimes. In bottomland hardwood ecosystems of south-central Louisiana, hens selected nest-sites in relation to forest canopy gaps and the resulting under-story growth similar to conditions observed in our study (Byrne and Chamberlain 2013). However, in this bottomland hardwood system, nest-site selection was heavily driven by an area's propensity to flooding. Nest-site selection in northeastern South Dakota, where both Rio Grande (*Meleagris gallopavo intermedia*) and Eastern subspecies were studied, was associated with higher numbers of trees and shrubs near the nest dissimilar to the Ozark Highlands (Leham et al. 2002). However, in more open landscapes these associations are likely driven by the low availability of vertical cover compared to the oak-hickory forests of the Ozark Highlands. Selection in Rio Grande hens along the South Platte River in northeastern Colorado was based on greater visual concealment, greater canopy cover, greater shrub cover, and lower grass cover (Schmutz et al. 1989). Merriam's hens (*Meleagris gallopavo merriami*) in a Douglas-fir (*Pseudotsuga menziesii*) ecosystem in Oregon selected nest-sites based on visual concealment and shrub densities, similar to the nest habitat selected by hens in our study (Lutz and Crawford 1987). Overall hens throughout North America select nest-sites based on a few common characteristics like visual concealment in an effort to maximize the probability that nest-site will be successful.

Ultimately, hens select nesting habitat to increase the probability of nest success. Many studies have found different factors that influence nest survival and success such as weather, habitat, and management practices (Badyeav 1994; Miller et al. 1998; Roberts et al. 1998; Thogmartin 1999; Moore et al. 2010; Ludwig 2012; Fuller et al. 2013). Our results, like many others, indicated that the greatest influence on nest survival was visual concealment (0-1 m) (Badyaev 1995; Moore et al. 2010; Fuller et al. 2013). Similar to Moore (1995) and Thogmartin (1999), we found that roads were an important part of the nesting process. However, we found nest-sites that were located closer to roads survived longer, unlike Thogmartin (1999) whom found wild turkey avoided nesting near roads. We also observed a linear trend that best explained nest survival in combination with visual concealment and distance to a road. This trend allowed for nest survival to decrease over the 28 day incubation period. Decreased survival over the incubation period could be a result of the buildup olfactory and visual predator ques. Thogmartin (2001) suggested that habitat quality reduces the effect of visual and olfactory cues to predators, and our results support this claim since higher levels of visual concealment and the location of nest-sites close to roads increase nest survival. Increase survival of nest-sites that are close to roads could be a result of habitat conditions created by increased light levels that reach the forest floor due to the canopy gaps created by roads. Roads could also provide better access for hens to and from their nests, resulting in the creation of few visual and olfactory predator cues. Also traffic along roads potentially creates olfactory and auditory disturbances that could mask any predatory cues that hens may create during incubation.

Our top model estimated nest success at 19% (SE = 5%) which is amongst the lowest reported across the wild turkey range. Our nest success rate was similar to that estimated by Badyaev (1995; 20% SE = 5%) before woodland and savanna restoration. Badyaev's (1995) and our estimates were lower than those reported elsewhere (Vangilder et al. 1987; Miller et al. 1998; Paisley et al. 1998) but not as low as reported by Paisley et al. (1998; 16%). Hen success is another metric used as an index of productivity for wild turkey. Hen success ranges from 0.0 to 100% with most falling between 30 and 70% (Vangilder 1992). Our adult hen success rate on WRERA (24.5%) was amongst the lowest reported for eastern wild turkey (19.5% reported by Thogmartin and Johnson 1999 – 82.2% reported by Vangilder 1992). Our hen success rate is lower than the recommended 40% for a high density wild turkey population (Kurzejeski et al. 1987).

Renesting rates have been found to be important factors in the maintenance of wild turkey populations, and were limited to only 21% in 2012 and 26% in 2013 on the WRERA (23.5% Mean; Vangilder 1992; Roberts and Porter 1996). Our renesting rate was lower than the 44% reported in Missouri by Vangilder et al. (1987) and the 38% reported by Badyaev (1994) on the WRERA. These estimates were documented in a similar region and in similar habitat to our study except for the presence of early growing season prescribed fire. Renesting has been suggested as an important component of wild turkey population dynamics that mitigate most potential negative impacts that early growing season prescribed fire might have on initial nest success (Williams 2012). Our observed decrease in renesting rate since the implementation of early growing season prescribed fire is cause for concern about the impacts of early growing season prescribed fire on the habitat resources necessary for renesting individuals. Since we observed no major differences in habitat characteristic and survival of renest-sites compared to initial nest-sites the renesting rate is likely tied to habitat conditions and available resources for hens to produce a second clutch. Currently insufficient information exits on the energetics of renesting in wild turkey and how those energetics are related to habitat quality especially in habitat managed with early growing season prescribed fire.

5 Conclusions

Our results show that hens selected nest-sites with high visual concealment, high percent slope, and more woody ground cover. Our findings currently do not clearly establish a relationship between early growing season prescribed fire for restoration and nest-site selection. However, as time since fire increases so does visual concealment (0-1 m), a critical factor in nest-site selection and linked to nest survival and success. This relationship being the case, we can conclude that the response of vegetation to prescribed fire does increase certain characteristics important in the selection of wild turkey nesting habitat. Nest success and hen success are both below recommended levels for high density populations. These estimates do not substantially differ from estimates collected before woodland and savanna restoration. In terms of productivity the similarity of estimates suggest the implementation of landscape scale prescribed fire for woodland and savanna restoration has not benefited wild turkeys on the WRERA. Our renesting rate has also decreased since the implementation of early growing since fire, and has not been related to an increase in initial nest success that would be expected in a population with a decreased renesting effort.

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Table 1. Model selection results for nest-site selection of eastern wild turkey from 2012 to 2013 in the Ozark Highlands, Arkansas, USA. *k* represents the number of estimated parameters in each model. AIC_c represents the corrected Akaike's Information Criterion for finite sample sizes. ΔAIC_c represents the relative difference in AIC_c to the top model. w_i represents the Akaike model weights. Models with ΔAIC_c <4 were retained along with the global and intercept models.

Candidate Models	*k*	AIC_c	ΔAICc	w_i
% Visual Concealment + Time Since Fire (yrs) + % Slope + % Woody Stem Ground Cover (GC) + % Visual Concealment*Time Since Fire (yrs)	7	204.5	0.00	0.60
% Visual Concealment + Distance to Roads (m) + Time Since Fire (yrs) + % Slope + % Woody Stem Ground Cover (GC) + % Grass GC + %Canopy Cover + % Visual Concealment*Time Since Fire (yrs)	10	206.6	2.09	0.20
Global	23	209.4	4.95	0.05
Intercept	2	297.49	93.01	0.00

Table 2. Model averaged parameter estimates, standard errors and lower and upper 95% confidence intervals for nest-site selection of initial nest attempts and renest attempts by wild turkey during 2011-2013 breeding seasons in the Ozark Highlands, Arkansas, USA.

Covariate	β	SE	95% CI Lower	95% CI Upper
Initial Nest Attempts				
% Visual Concealment (0-1m)	4.24	1.00	2.26	6.2
%Visual Concealment (0-1m)*Time Since Fire (yrs)	3.81	1.71	0.46	7.17
% Slope	1.85	0.62	0.63	3.07
% Woody Stem Ground Cover	1.36	0.58	0.23	2.51
Renest Attempts				
% Visual Concealment (0-1m)	2.12	0.96	0.23	4.01
Count of Small Shrubs	-3.48	1.57	-6.55	-0.39

Table 3. Model selection results for renest-site selection of eastern wild turkey from 2012 to 2013 in the Ozark Highlands, Arkansas, USA. *k* represents the number of estimated parameters in each model. AIC_c represents the corrected Akaike's Information Criterion for finite sample sizes. ΔAIC_c represents the relative difference in AIC_c to the top model. w_i represents the Akaike model weights. Models with ΔAIC_c <4 were retained along with the global and intercept models.

Model	*k*	AIC_c	ΔAIC_c	w_i
% Visual Concealment (0-1m) + Distance to Road (m) + Count of Medium Trees + Count of Small Shrubs	6	55.03	0.00	0.34
% Visual Concealment (0-1m) + Distance to Road (m) + Count of Medium Trees + Count of Small Shrubs + Time Since Fire (yrs)	7	55.72	0.69	0.24
% Visual Concealment (0-1m) + Distance to Road (m) + Count of Medium Trees + Count of Medium Trees*Time Since Fire (yrs) + Count of Small Shrubs	7	57.33	2.31	0.11
% Visual Concealment (0-1m) + Distance to Road (m) + Count of Medium Trees + Count of Small Shrubs + Count of Small Shrubs*Time Since Fire (yrs)	7	57.58	2.55	0.09
% Visual Concealment (0-1m)	3	57.84	2.81	0.07
% Visual Concealment (0-1m) + Distance to Road (m)	5	58.59	3.56	0.06
Intercept	2	59.59	4.56	0.03
Global	23	94.96	39.93	0.00

Table 4. Model selection results for daily nest survival analysis of eastern wild turkey from 2012 to 2013 in the Ozark Highlands, Arkansas, USA. *k* represents the number of estimated parameters in each model. AIC_c represents the corrected Akaike's Information Criterion for finite sample sizes. ΔAIC_c represents the relative difference in AIC_c to the top model. w_i represents the Akaike model weights.

Model	*k*	AIC_c	ΔAIC_c	w_i
S(T + Percent Visual Concealment (0-1m) + Distance from Road (m))	3	397.0	0.00	0.65
S(null model)	1	409.6	12.7	0.00

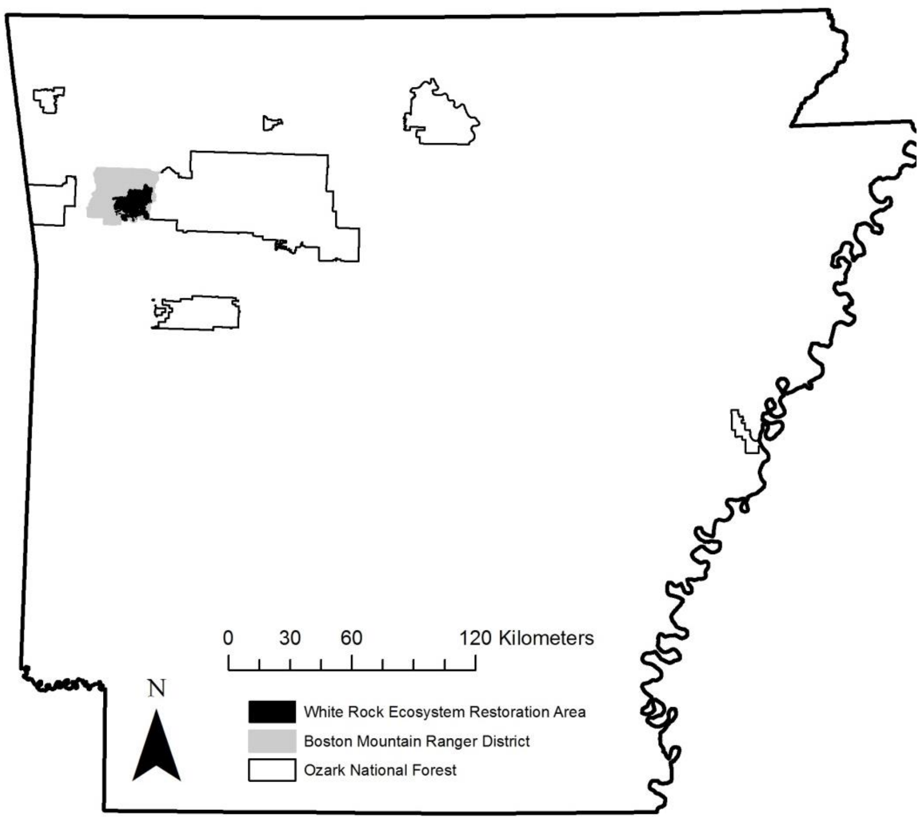

Fig 1. White Rock Ecosystem Restoration Area, Boston Mountain Ranger District, and the Ozark National Forest.

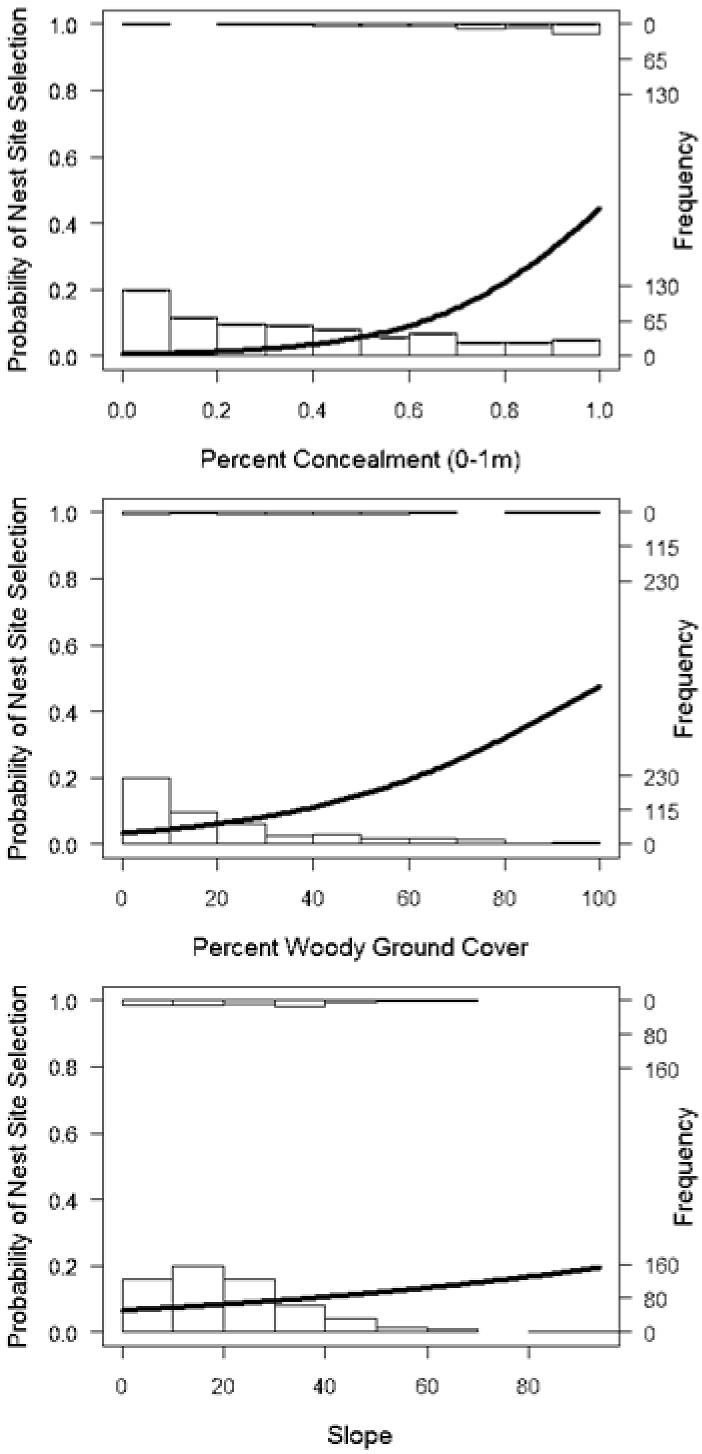

Fig 2. Logistic regression curves for model averaged non-zero variables with frequency distributions of variable values for unused (Bottom) and nest (Top) sites of wild turkey from 2011 to 2013 in the Ozark Highlands, Arkansas, USA.

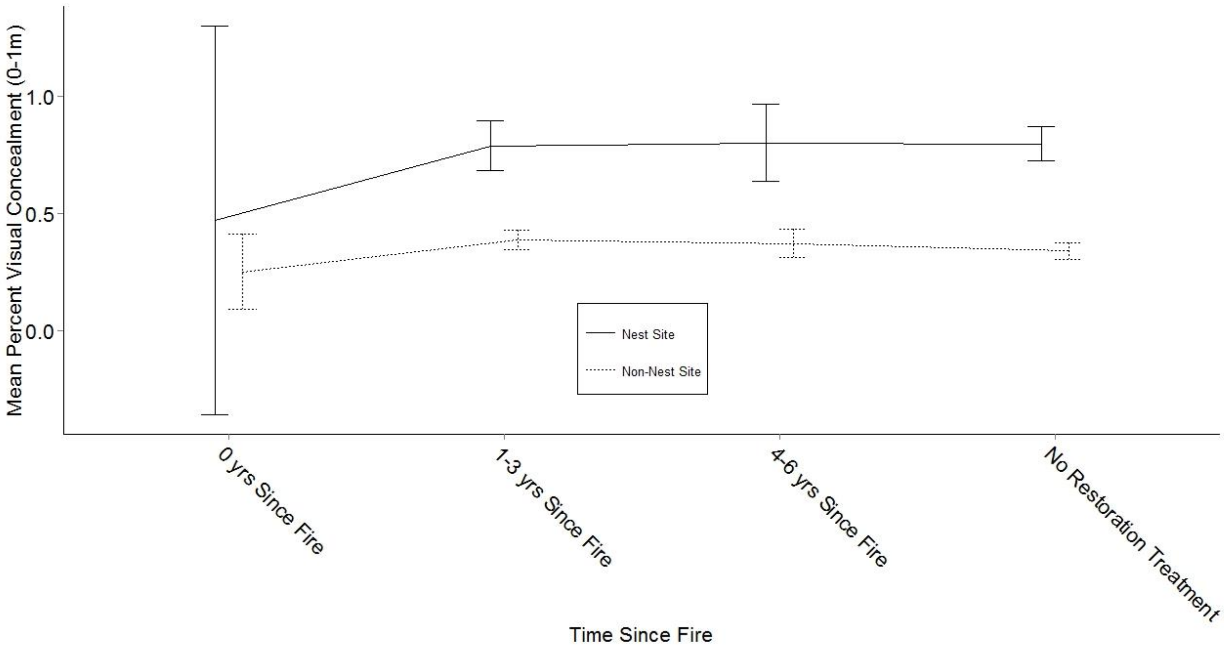

Fig 3. Interaction relationship between percent visual concealment (0-1m) and time since fire for initial nest attempts and unused locations from 2011 to 2013 in the Ozark Highlands, Arkansas, USA.

Appendix A. Full model selection results for initial nest-site selection analysis of eastern wild turkey in the Ozark Highlands, Arkansas, USA. *k* represents the number of estimated parameters in each model. AIC_c represents the corrected Akaike's Information Criterion for finite sample sizes. ΔAIC_c represents the relative difference in AIC_c to the top model. w_i represents the Akaike model weights.

Model	*k*	AIC_c	ΔAIC_c	w_i
% Visual Concealment + Time Since Fire (yrs) + % Slope + % Woody Stem Ground Cover (GC) + % Visual Concealment*Time Since Fire (yrs)	7	204.48	0.00	0.58
% Visual Concealment + Distance to Roads (m) + Time Since Fire (yrs) + % Slope + % Woody Stem Ground Cover (GC) + % Grass GC + % Canopy Cover + % Visual Concealment*Time Since Fire (yrs)	10	206.57	2.09	0.20
Global (% Visual Concealment + Distance to Roads (m) + Time Since Fire (yrs) + % Slope + % Woody Stem Ground Cover (GC) + % Grass GC + %Canopy Cover + Count of Medium Trees + Count of Small Shrubs + Slope Position + Aspect + % Visual Concealment*Time Since Fire (yrs) + % Woody Stem Ground Cover*Time Since Fire (yrs) + % Woody Stem Ground Cover:Time Since Fire (yrs) + Count of Medium Trees:Time Since Fire (yrs))	23	209.422	5.03	0.05
% Visual Concealment + Distance to Roads (m) + Time Since Fire (yrs) + Count of Medium Trees + Count of Small Shrubs + Count of Small Shrubs*Time Since Fire(yrs)	7	209.5	5.02	0.05
% Visual Concealment + Distance to Roads (m) + Time Since Fire (yrs) + % Woody Stem GC + % Slope + % Woody Stem GC*Time Since Fire (yrs)	8	209.58	5.11	0.05
% Visual Concealment + Distance to Roads (m) + Time Since Fire (yrs) + % Woody Stem GC + % Slope + % Grass GC + % Canopy Cover	9	209.98	5.51	0.04
% Visual Concealment + Distance to Roads (m) + Count of Medium Trees + Count of Small Shrubs	6	212.56	8.08	0.01
% Visual Concealment + Distance to Roads (m) + Time Since Fire (yrs) + Count of Medium Trees + Count of Small Shrubs	7	212.7	8.22	0.01

Appendix A Cont.

Model	k	AIC_c	ΔAIC_c	w_i
% Visual Concealment + Distance to Roads (m) + % Woody Stem GC + % Grass GC + % Canopy Cover	7	213.67	9.2	0.01
% Visual Concealment + Distance to Roads (m) + Time Since Fire (yrs) + % Slope + Slope Position	8	214.11	9.64	0.00
% Visual Concealment + Distance to Roads (m) + Time Since Fire (yrs) + Count of Medium Trees + Count of Small Shrubs + Count of Medium Trees*Time Since Fire(yrs)	7	214.47	10.0	0.00
% Visual Concealment + Distance to Roads (m)	4	216.04	11.56	0.00
% Visual Concealment + Distance to Roads (m) + Time Since Fire (yrs)	5	216.91	12.43	0.00
% Visual Concealment	3	218.16	13.69	0.00
Distance to Roads (m) + Time Since Fire (yrs) + % Woody Stem GC + % Grass GC + % Canopy Cover + Count of Medium Trees + Count of Small Shrubs	9	261.87	57.39	0.00
Distance to Roads (m) + Time Since Fire (yrs) + % Woody Stem GC + % Grass GC + Count of Medium Trees + Count of Small Shrubs	8	262.74	58.27	0.00
Intercept	2	297.49	93.01	0.00
Near Treatment Type	4	299.1	91.09*	0.00(0.35*)
Time Since Fire (yrs) + Stand Age + Stand Type	7	299.4	1.87*	0.16*
Stand type + Distance to Treatment (m) + Stand Age (yrs)	7	299.7	2.22*	0.14*
Stand type + Distance to Treatment (m) + Stand Age (yrs) + Treatment Age (yrs)	8	300.2	2.72*	0.12*

Appendix A Cont.

Model	k	AIC_c	ΔAIC_c	w_i
Time Since Fire (yrs) + Stand type + Distance to Treatment (m) + Stand Age (yrs)	8	301.3	3.84*	0.05*
Stand type + Stand Age (yrs) + Near Treatment Type + Treatment Age (yrs) + Treatment Area	10	301.4	3.96*	0.05*
Near Treatment Type + Treatment Age (yrs) + Treatment Area + Distance to Treatment (m)	7	302.8	5.34*	0.02*
Stand type + Stand Age (yrs) + Near Treatment Type + Treatment Age (yrs) + Treatment Area + Distance to Treatment (m)	77	303.1	5.58*	0.02*

*Management candidate model compared to the intercept only model

Appendix B. Full model selection results for daily nest survival analysis of eastern wild turkey in the Ozark Highlands, Arkansas, USA. *k* represents the number of estimated parameters in each model. AIC_c represents the corrected Akaike's Information Criterion for finite sample sizes. ΔAIC_c represents the relative difference in AIC_c to the top model. w_i represents the Akaike model weights.

Model	k	AIC_c	ΔAIC_c
S(Temporal Trend+% Visual Concealment + Distance from Road)	396.9	0.0	0.57
S(Temporal Trend+% Visual Concealment)	399.3	2.4	0.17
S(Temporal Trend+% Visual Concealment + Time Since Fire)	401.3	4.4	0.06
S(Temporal Trend + Dispersal Status)	402.0	5.1	0.05
S(Temporal Trend + Nest Attempt)	402.0	5.1	0.05
S(Temporal Trend)	402.8	5.9	0.03
S(Temporal Trend + Year)	402.8	5.9	0.03
S(Temporal Trend+% Visual Concealment+% Woody Ground Cover + No. of Medium Trees+% Slope)	403.2	6.3	0.02
S(Temporal Trend + Canopy Cover + Understory Height + No. of Small shrubs)	405.1	8.2	0.01
S(% Visual Concealment + Distance from Road)	407.2	10.4	0.00
S(% Visual Concealment)	408.0	11.1	0.00
S(Distance from Road)	408.9	12.0	0.00
S(null)	409.6	12.7	0.00
S(Dispersal Status)	409.6	12.7	0.00
S(Nest Attempt)	409.7	12.8	0.00
S(Year)	410.2	13.3	0.00
S(t)	419.5	22.6	0.00

Appendix C. Model selection results from the pre-nesting discrete choice habitat selections analysis. Also included in *italics* are specific hypotheses that the models represent. Interpretability was not advisable since model fit was not sufficient for all models.

Model	k	AIC_c	ΔAIC_c	w_i
Global	27	317.10	29.16	0.00
Nest-Site Selection				
% Visual Concealment + Distance to Roads (m) + Time Since Fire (yrs) + Count of Medium Trees + Count of Small Shrubs	5	294.45	6.51	0.02
% Visual Concealment + Distance to Roads (m) + Time Since Fire (yrs) + % Slope + % Woody Stem GC + % Grass GC + % Canopy Cover	7	296.32	8.38	0.01
% Visual Concealment + Distance to Roads (m) + Time Since Fire (yrs) + % Slope + Slope Position + Aspect	12	300.67	12.73	0.00
Predator Avoidance				
Count of Large Trees + Count of Small Trees + Understory Height (cm) + % Slope + Slope Position	5	291.58	3.64	0.08
Count of Large Trees + Understory Height (cm) + % Litter GC + % Canopy Cover	4	289.71	1.77	0.20
Count of Large Trees + Time Since Fire (yrs) + Understory Height (cm) + Slope Position + Aspect	11	299.25	11.31	0.00
Basal Area Hardwoods + Basal Area Conifers + Understory Height (cm)	3	293.34	5.40	0.03
Travel Corridors				
Distance to Roads (m) + Slope Position	2	291.33	3.39	0.09
Nutrients and Energy				
Distance to Treatment (m) + Nearest Treatment Age + % Grass GC + % Litter GC	4	295.49	7.55	0.01
Distance to Roads (m) + % Grass GC + % Litter GC + % Canopy Cover	4	294.09	6.15	0.02
Time Since Fire (yrs) + % Grass GC + % Canopy Cover	3	292.62	4.68	0.05
Stand Age (yrs) + Site Index	2	287.94	0.00	0.49

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MEMORANDUM

TO: David Krementz

FROM: Craig N. Coon, Chairman
Institutional Animal Care
And Use Committee

DATE: November 9, 2010

SUBJECT: IACUC PROTOCOL **APPROVAL**
Expiration date : **November 8, 2013**

The Institutional Animal Care and Use Committee (IACUC) has **APPROVED** Protocol **#11012- " EFFECTS OF LARGE SCALE GROWING SEASON PRESCRIBED BURNS ON MOVEMENTS, HABITAT USE, PRODUCTIVITY, AND SURVIVAL OF FEMALE WILD TURKEYS ON THE WHITE ROCK ECOSYSTEM RESTORATION PROJECT OF THE OZARK-ST. FRANCIS NATIONAL FOREST "**. You may begin this study immediately.

In granting its approval, the IACUC has approved only the protocol provided. Should there be any changes in the protocol during the research, please notify the IACUC in writing **prior** to initiating the changes. If the study period is expected to extend beyond, **11-08-2013** you must submit a new protocol. By policy the IACUC cannot approve a study for more than 3 years at a time.

The IACUC appreciates your cooperation in complying with University and Federal guidelines for research involving animal subjects.

cnc/car

cc: Animal Welfare Veterinarian

J. William Fulbright College of Arts and Sciences
Department of Biological Sciences

Chapter 1, "Impacts of upland hardwood ecosystem restoration through early growing season prescribed fire on Eastern Wild Turkey Nest Ecology in the Ozark Highlands, Arkansas, USA" of H. T. Pittman's dissertation is intended for submission for publication with one coauthor, D. G. Krementz.

I, Dr. David G. Krementz, advisor of Henry Tyler Pittman, confirm Henry Tyler Pittman will be first author and completed at least 51% of the work for this manuscript.

David G. Krementz
Unit Leader
U. S. Geological Survey
Arkansas Cooperative Fish and Wildlife
Research Unit

Date

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Chapter II

Impacts of Landscape Level Woodland and Savanna Restoration on Eastern Wild Turkey Movement and Reproductive Ecology

Henry Tyler Pittman

And

Dr. David G. Krementz

Abstract

Landscape level woodland and savanna restoration is becoming an increasingly common management strategy in the Central Hardwoods region. The implementation of the restoration strategy has coincided with declines of Eastern Wild Turkey in many treated areas, especially on the White Rock Ecosystem Restoration Area (WRERA). Previous studies have documented that habitat quality is related to individual movement and can limit a population. We fitted 67 female wild turkey with 110 g Global Positioning System (GPS) Platform Transmitting Terminals in 2012 and 2013 on the WRERA to estimate annual and seasonal home ranges and examine pre-incubation habitat use at multiple spatial scales. We estimated mean 95% home ranges for adults as 2,703 ha (SE=283.4) in 2012 and 4,750.4 ha (SE=946.2) in 2013 and for sub-adults as 1,999.6 ha (SE=440.5) in 2012 and 4,169.5 ha (SE=751.7) in 2013. We estimated mean 50% core use areas of adults as 387.7 ha (SE=29.4) in 2012 and 373.3 ha (SE=38.2) in 2013 and for sub-adults as 355.2 ha (SE=74.5) in 2012 and 358.2 ha (SE=92) in 2013. Compared to before restoration, pre-incubation ranges were similar for adults but were larger for sub-adults and home ranges for all hens were larger after restoration. Our habitat selection analyses for the pre-incubation period indicated that hens visited habitats with more structural diversity and in smaller patch sizes than available habitat, but that hens eventually selected nesting habitat that had greater structural diversity in larger patches.

Introduction

Landscape level woodland and savanna ecosystem restoration is a prominent forest management strategy implemented across the Central Hardwoods region of North America. The aim of woodland and savanna restoration management is to return closed canopy upland forests to their historical structure and functions. Historical conditions on the western edge of the Central Hardwoods region consisted of woodlands and savannas that represented a transition zone between tall grass prairie in the Central Plains and closed canopy deciduous forest in eastern North America (Nuzzo 1986). Woodland and savanna ecosystems consisted of open canopies with diverse understories of grasses, forbs, and some woody vegetation and were maintained through disturbance, typically fire (McPherson 1997; Taft 1997; Anderson et al. 1999). Swan (1970) reported these woodlands were more stable, had better oak regeneration and maintained more herbaceous vegetation. On the western edge of the Central Hardwoods, restoration is implemented through prescribed fire and mechanical reductions in canopy cover across large areas resulting in woodland or savanna conditions.

Woodland and savanna restoration techniques, especially prescribed fire, have been examined for their impacts and effectiveness on forest structure at multiple spatial scales (Hutchinson et al. 2005; Albrecht and McCarthy 2006; Brose et al. 1999). Various implementation techniques and impacts of prescribed fire on forest structure have been studied including the reintroduction, season, intensity, and effectiveness as a system maintaining disturbance (Sparks et al. 1998; Peterson and Reich 2001; Hutchinson et al. 2005; Franklin et al. 2003). Prescribed fire has also been examined and found most effective for woodland and savanna restoration when implemented in combination with mechanical canopy reductions and herbicide treatments (Hutchinson et al. 2005; Albrecht and McCarthy 2006; Brose et al. 1999;

Jackson and Buckley 2004). Research has also focused on the effects that woodland and savanna restoration techniques have on wildlife (Brawn et al. 2001; Davis et al. 2000; Teidemann et al. 2000; Reidy et al. 2014). However, to our knowledge, no study has examined the impacts of prescribed fire alone as a woodland and savanna restoration tool at the landscape scale (≥10,000 ha) at which it is currently implemented has on wildlife.

In 2002, woodland and savanna restoration was initiated in the Ozark Highlands of Arkansas, USA on a landscape scale (≥10,000 ha) in an effort to return closed canopy forest to historic woodland and savanna conditions. Harvest of eastern wild turkey (*Meleagris gallopavo silvestris*) in the Ozark Highlands reached an all-time highest level in 2003, but harvest has since declined to the lowest levels since 1991 when newly introduced wild turkey populations were still small (K. Lynch, Arkansas Game and Fish Commission, unpub. data). Wild turkey harvest is often used to index population status and these declines led to concerns that changes in forest management were affecting hen productivity. Space use and habitat relationships of hens have been well documented during the critical reproductive period under numerous forest management scenarios (Lazarus and Porter 1985; Badyaev 1994; Thogmartin 2001; Miller and Conner 2007). Although hens use a variety of habitats during this demographically important reproductive period, they require specific habitat characteristics during nesting related to forest structure such as visual concealment (Badyaev 1994; Thogmartin 2001; Byrne and Chamberlain 2013). Thogmartin (1999) hypothesized that the availability of suitable nest-sites may be a factor limiting population size. Hens also expend large amounts of energy in the search of nesting habitat which can ultimately impact reproductive fitness (Badyaev 1996; Thogmartin 2001; Miller and Conner 2005). Recognizing the relationships between wild turkey nest ecology and woodland restoration techniques led to concerns that current woodland and savanna restoration efforts were adversely affecting wild turkey populations.

Badyaev (1996) first suggested that pre-incubation movements of hens were related to the selection of nest-sites and the success of those nests. Other researchers have also found that the pre-incubation period was critical in nest-site selection and nest success (Thogmartin 2001; Chamberlain and Leopold 2000; Miller and Conner 2005). Thogmartin (2001) suggested that hens would even trade optimal foraging habitat in search of ideal nesting habitat, indicating the importance of habitat sampling in the nest-site selection process. Other research has also connected pre-incubation home range size to the quantity and availability of quality nesting habitat suggesting search area and intensity should decrease as landscape habitat quality increases (Stephens and Krebs 1986; Orain and Wittenberger 1991; Chamberlain and Leopold 2000). Miller and Conner (2007) examined pre-incubation habitat selection, but their assessment only described habitat of hens' pre-incubation home ranges, and not the sequential habitat selection process a hen undergoes while moving through the landscape in search of quality nesting habitat.

We initiated this study to better understand the effects of woodland and savanna restoration on hen reproductive ecology. Our objectives were to: 1) document home ranges and pre-incubation ranges for hens under landscape level woodland and savanna restoration conditions, 2) investigate potential difference between these ranges and ranges documented before the implementation of restoration, 3) determine how current pre-incubation ranges relate to productivity, 4) examine pre-incubation habitat sampling at multiple spatial scales, and 5) examine how habitat sampling during the pre-incubation period relates to eventual habitat selected for nest-sites. Through this study we hope to better understand how hens use the landscape during the pre-incubation period and to understand what impacts landscape level forest management may have on hen reproductive ecology.

Methods

Study area

The White Rock Ecosystem Restoration Area (WRERA) consists of 16,380 ha of upland hardwood and pine ecosystems in Northwest Arkansas, United States of America (USA; Figure 1). It is part of the larger main division of the Boston Mountain Ranger District (41,400 ha) on the Ozark-St. Francis National Forest. The WRERA is a high priority woodland and savanna restoration area for the United States Department of Agriculture (USDA) Forest Service. Historically the WRERA was dominated by woodlands and savannas maintained by frequent disturbances by fire (Cutter and Guyette 1994; Batek et al. 1999; Foti 2004; Chapman et al. 2006; Guyette et al. 2006; Stambaugh and Guyette 2006). After fire suppression during the 20th century, the WRERA became dominated by closed canopy hardwood forests of various oak (*Quercus sp.*) and hickory (*Carya sp.*) species. Understories consisted of canopy species regeneration, black gum (*Nyssa sylvatica*), flowering dogwood (*Cornus florida*), Carolina buckthorn (*Frangula caroliniana*), black berry (*Rubus spp.*) and devil's walking stick (*Aralia spinosa*) (USDA, NRCS 2014). Pine ecosystem canopies were dominated by short leaf pine (*Pinus echinata*) while understories consisted of hardwood and pine regeneration. Since 2002 the WRERA has been intensively managed with large scale early growing season prescribed fire to restore historic woodland and savanna conditions. The management prescription for WRERA describes woodlands as having "open canopies, sparse mid-stories and well-developed understories that are typically dominated by grasses and forbs, but also may become shrubby between fires and have a significant woody component" with 40 to 60% canopy closure (Ozark NF Plan 2005: pp. F-7). The prescription also describes the management techniques to achieve these objectives including mechanical canopy removal, herbicide, and fire treatments. However, mechanical and herbicide treatments have been mostly absent in the restoration area since 2002. The WRERA is divided into 16 prescribed fire units ranging from 467 to 1,670 ha in size. Early growing season prescribed fire has been implemented on a 3 to 5 year rotation in 15 of the 16 prescribed fire units, and all units received 1 to 4 prescribed fire treatments since 2002.

Capture and GPS telemetry

We trapped hens in and around the WRERA using rocket nets in late winter from 2012-2013 (IACUC #11012). All hens were fitted with a 110 gram Platform Transmitting Terminal (PTT) with Global Positioning System (GPS) capabilities and Very High Frequency (VHF) radio transmitter (North Star Science and Technology LLC, King George, Virginia) attached using a back-pack-type harness (Kurzejeski et al. 1987). The PTTs recorded GPS locations four times per day during the 2012 nesting seasons and eight times per day during the 2013 nesting season. We estimated GPS location accuracy to approximately 15 m. Our PTTs transmitted GPS locations via the ARGOS satellite system (CLS America) every 120 hrs during 2012 and every 48 hrs during 2013.

Home range estimation

We estimated home and seasonal ranges from GPS coordinates using three techniques, dynamic Brownian Bridge Movement Models (dBBMM) (Kranstauber et al. 2012,2013; R package="*move*"), fixed kernels (kernel), and minimum convex polygons (MCP) (Calenge 2006; R package="*adehabitat*"). We used estimates derived from the dBBMM, but we report estimates from all three methods in figures, and we report appropriate method estimates for range comparisons (Laver and Kelly 2008). We made range comparisons through visual observation of plots of means, box and whisker plots, standard errors and 95% confidence intervals (95% CI). We defined an individual's home range as a continuous period of locations ranging from 10 to 12 months that incorporated at least 50% of all seasonal behaviors, pre-incubation, nesting, brood rearing, summer-fall, and winter. We defined the pre-incubation period separately for every individual since individuals did not leave their winter flocks at the same time. The pre-incubation period began when a bird appeared to separate from others in its winter flock and ended two weeks before the beginning of nest incubation. We defined the nest incubation as the point when at least four consecutive GPS locations were within 25 m of each other. The pre-incubation period ranged from early March to early April in 2012 and from mid-March to late April in 2013.

Habitat selection

We estimated habitat selection for hens during the pre-incubation period using the discrete choice model outlined by Cooper and Millspaugh (1999). We estimated the discrete choice model (Croissant 2011; R package="*mlogit*") at multiple spatial scales, point, 100 m, 200 m and 400 m scales, since hens have been documented to select habitat in a hierarchical fashion (Lazarus and Porter 1985; Byrne and Chamerlain 2013). We also estimated nest-site selection using the discrete choice model to compare eventual nesting habitat selected to habitat sampled during the pre-incubation period. We used Akaike's Information Criterion corrected for finite sample size (AIC_c) to rank candidate models (Burnham and Anderson 2002). We only report model selection results for spatial scales that produced at least one candidate model with $\Delta AIC_c > 2$ of full model. We performed all statistical modeling and analyses using R statistical language (R Core Development Team, 2014).

We used dBBMM segment 95% contours (Collier 2013; R package=”*moveud*”) to define unique choice sets for every choice of each hen to better understand the selection process of a hen as it moves across the landscape and samples available habitat. We then generated a set of 10 available random point locations within each segment 95% contour to use as alternatives in the choice set. We combined the 10 available alternatives with the second location from the pair that composed the segment resulting in choice sets of 11 alternatives. The second location from the pair that composed the segment was used as the chosen alternative since the individual was known to be at that location. Our choice sets were based on the premise that an individual can choose among available locations, including its current location, depending on which maximize the utility to that individual.

We extracted data from multiple continuous landscape level data sets for every alternative at each scale including our nest-site selection locations. At the point scale, we used 2010 LANDFIRE vegetation datasets (Rollins 2009) for canopy cover and vegetation height and 2011 National Land Cover Datasets (NLCD) for forest cover type (Rollins 2009; Homer et al. 2007; Jin et al 2013). We used the raster calculator and focal statistics tools (ESRI 2013) to develop datasets for percent change in canopy cover from 2001 to 2010, canopy cover heterogeneity in 2010, change in vegetation height from 2001 to 2010, vegetation height heterogeneity in 2010, and NLCD cover type heterogeneity for 2011. At larger scales we generated buffers of 100, 200, and 400 m, around the point locations and calculated the mean of each vegetation variable within that area, except for NLCD cover type. For NLCD cover type at each scale, we estimated the percentage of the area of each buffer composed of the four most dominant cover types in the study area, evergreen forest, deciduous forest, grassland/herbaceous, and pasture/hay. We then z-score standardized all of the variables at all scales, except NLCD cover type at the point scale since it was categorical. We developed our candidate model sets separately for all scales and analyses. Our candidate models were based on findings of Badyaev (1994), Thogmartin (1999), Miller and Conner (2005), and Byrne and Chamberlain (2013). Our hypotheses about disturbance and forest management practices are available upon request.

Results

Home range estimation

We estimated home ranges for 38 hens from 2012 and 2013 on WRERA. We observed large mean 95% home ranges for adults, 2,703 ha (SE=283.4) in 2012 and 4,750.4 ha (SE=946.2) in 2013. Sub-adult 95% home ranges were 1,999.6 ha (SE=440.5) in 2012 and 4,169.5 ha (SE=751.7) in 2013. These home range estimates were considerably larger than mean home range estimates previously documented in Arkansas (Table 1). We estimated mean 50% core use areas for adults, 387.7 ha (SE=29.4) and 373.3 ha (SE=38.2) in 2012 and 2013 respectively, and for sub-adults, 355.2 ha (SE=74.5) and 358.2 ha (SE=92) in 2012 and 2013 respectively. We found a positive relationship between nest success and pre-incubation range size in 2013 when dBBMM pre-incubation ranges on average were larger for successfully nesting hens than those that were unsuccessful and the reversed in 2013 (Figure 2). However 95% confidence intervals overlapped between successful and unsuccessful hens and between years. We also examined this relationship across fixed kernel and MCP estimates of pre-incubation ranges and found there was too much overlap in the distributions of pre-incubation ranges to suggest any differences (Figure 3). We found no relationship between dBBMM estimates of 95% home ranges and 50% core use areas for successful and unsuccessful hens.

Due to the variation between methods observed for successful and unsuccessful hens we used all methods for examination of year and age effects on range sizes. We found dBBMM estimates were most similar to fixed kernel and MCP methods when comparing 95% home ranges and 50% core use areas for adults (Figure 4). However when adult pre-incubation range and sub-adults range estimates were compared, considerable variation existed between methods. This being the case, dBBMM estimates were most conservative and best accounted for outliers that may have resulted from variation in individual behavior (Figure 4). dBBMM estimates also indicated adult and sub-adult 95% home range and 50% core use area sizes were similar, but sub-adult pre-incubation range sizes were considerably larger than that of adults. When we compared estimation methods for year affects dBBMM estimates were most conservative and accounted for outliers the best in 95% home range and 50% core use area estimates. dBBMM estimates also suggested that no difference between years exist for 95% home ranges since a majority of their range distributions overlapped (Figure 5). Comparison of estimation method for pre-incubation ranges between years suggested that MCP estimates were most conservative, but in all estimates we observed considerable overlap of the range distributions suggesting no difference between years (Figure 5).

Habitat selection

Between 2011 and 2013, hens during the pre-incubation period were making habitat selection choices that were best differentiated from available alternatives at the point and 200 m scales. At the point scale, the top models selected were based on the use of early successional habitat and habitat structure diversity that would result in understory variation and still provide vertical cover. At the point scale, hens selected habitat with increasing canopy cover from 2001 to 2010 ($\sum w_i$=0.34), increasing vegetation height from 2001 to 2010 ($\sum w_i$=0.34), higher values of canopy cover ($\sum w_i$=0.48), and high canopy cover heterogeneity ($\sum w_i$=0.33; Table 2). At the 200 m scale hens selected habitat with higher values of canopy cover ($\sum w_i$=0.89) indicating the selection of habitat with vertical cover. Habitat selected for the eventual nest-site was best differentiated from available alternatives at the point, 100 m, and 200 m scales. At the point scale hens selected nest-sites that were best characterized by high canopy cover heterogeneity ($\sum w_i$=0.57) and lower canopy cover ($\sum w_i$=0.39; Table 3). At the 100 m and 200 m scales hen nest-sites were best differentiated from alternative sites by higher mean canopy cover heterogeneity (Table 3). When we compared these results between pre-incubation and nest-site selection, we found high canopy cover heterogeneity at the point scale for both selection periods. Among all scales, we found that canopy cover, change in canopy cover and vegetation height from 2001 to 2010, and canopy cover heterogeneity were all involved in the habitat sampling and selection process.

Discussion

Space use has been previously used as an indicator of habitat quality suggesting that increased space use is a result of a lack of quality habitat (Stephens and Krebs 1986; Cobb and Doerr 1997; Chamberlain and Leopold 2000; Wilson et al. 2005). Because we observed >100% increase in mean 95% home range size compared to the estimates before woodland and savanna restoration (Badyaev et al. 1996), our 95% home range estimates suggest that habitat quality has declined in the study area since the implementation of woodland and savanna restoration. Our estimates of home range size are also larger than others reported for eastern wild turkey (Everett et al. 1979; Badyaev et al. 1996; Thogmartin 2001; Miller and Conner 2005; Wilson et al. 2005). The large size of home ranges compared to other populations in other ecosystems and under different forest management conditions suggest a lack of quality habitat for hens on the WRERA compared to areas under different forest management scenarios.

Our estimates of pre-incubation range sizes also suggest a lack of quality habitat in the WRERA since the pre-incubation range of a hen in our study represents on average 46% of its total home range area. We also observed larger mean pre-incubation ranges in sub-adults than in adults which possibly suggest that dominant adults were selecting quality nesting habitat on WRERA forcing sub-adults to search for quality nesting habitat across a larger area. Badyaev et al. (1996) suggested a similar relationship but did not observe as large of a difference between the pre-incubation ranges of adults and sub-adults. Our results also suggest a difference in pre-incubation ranges between 2012 and 2013 that could potentially be due to environmental factors such as the effects of climate on habitat quality. Vegetation data collected over this period in the WRERA suggest that nesting habitat conditions may have been poor in 2013 because decreases in small shrubs and woody ground cover observed in 2013 could have been a result of severe drought in the summer of 2012. However, overlap of the distribution of all range estimates across all methods and susceptibility of mean ranges to outliers suggest caution when inferring differences between groups. The high variability within groups and methods also raises questions of the magnitude of trends and differences in other studies of hen home range estimates. In either event, large pre-incubation ranges that represent a major proportion of the largest documented annual home ranges for the sub-species suggest a lack of quality nesting habitat. A lack of quality nesting habitat, supported by the 19% nest success rate documented during this study, could be limiting the wild turkey population on WRERA (Thogmartin 2001).

Our pre-incubation habitat selection results suggest that hens are using more diverse habitats that may be in a transitional state (i.e. increasing canopy cover and vegetation height from 2001 to 2010). Hens use more diverse habitat and interspersion of habitat with differing structures during nest-site selection (Speake et al. 1975; Brown 1980; Miller and Conner 2005). Miller and Conner (2007) found the pre-incubation period was a time when hens transition between structurally different habitats that provided habitat characteristic for the reproductive and non-reproductive seasons. We also believe that during this period hens are searching for structurally diverse habitats because they often provide open canopy habitats interspersed with opportunities for visual concealment. The occurrence of structural variables such as canopy cover heterogeneity and change in canopy cover and vegetation height support this hypothesis. However, the occurrence of these variables at the point scale and absence of them at the 200 m scale suggests that hens are finding these structurally diverse habitats in relatively small patches. Being smaller in size, we believe that the habitats may be gaps created by disturbances such as fire, wind or insects. The eventual nest-sites that hens select suggest they are ultimately looking for structural diversity. Structural diversity is indicated by the occurrence of canopy cover heterogeneity in top models at the point and 100 m scales of our nest-site selection analysis, similar to the results of the pre-incubation habitat selection analysis. However, the occurrence of higher mean canopy cover heterogeneity at the 100 m scale and a reduction in vegetation height since 2001 at the 200 m scale suggest hens ultimately select nest-sites in larger patches with structural diversity. At the 100 m and 200 m scales these patches are likely created by large disturbances such as large insect infestations or severe weather events. Disturbances like these are limited in the WRERA, occurring on $\leq$5% of the 16,380 ha. Badyaev (1994) and Thogmartin (1999) both found that hens selected large nesting patches similar to what we observed. Byrne and Chamberlain (2013) also found selection of nest-site patches at the same 200 m scale, but found that at larger scales habitat diversity was negatively related to nest-site selection.

In summary, our results suggest that a lack of quality habitat is the cause of the large home and seasonal ranges of hens on WRERA. Our results indicated that this more specifically was due to a lack of quality nesting habitat causing increased habitat sampling during the pre-incubation period especially in sub-adults. We found that woodland and savanna restoration specifically landscape level prescribed fire has not increased the amount of quality nesting habitat. Our habitat selection results indicate that hens are looking for structurally diverse habitats. Hens on the WRERA are eventually selecting large patches of structurally diverse habitat likely created by larger more severe canopy disturbances.

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Table 1 Mean [SE] Home range estimates for hens in 2012 and 2013 on the White Rock Ecosystem Restoration Area (WRERA), Boston Mountain Ranger District of the Ozark-St. Francis National Forest, Arkansas, USA and estimates from Thogmartin (1999) in the Ouachita Highlands of Arkansas from 1993 to 1996 and Badyaev and Faust (1996) on the WRERA from 1992 to 1993 before woodland and savanna restoration

Adults	50% Kernels	95% Kernels
2012	387.7 [29.4]	2703 [283.4]
2013	373.3 [38.2]	4750.4 [946.2]
Badyaev and Faust 1996	-	1414.3[A]
Thogmartin 2001	-	1630[B]
Sub-adults		
2012	355.2 [74.5]	1999.6 [440.5]
2013	358.2 [92]	4169.5 [751.7]
Badyaev and Faust 1996	-	3929.2[A]
Thogmartin 2001	-	3200[B]

[A]90% Minimum Convex Polygon Home Range Estimate

[B]95% Fixed Kernel Home Range Estimate

Table 2 Discrete choice model selection results during the pre-incubation period for hens from 2012 and 2013 on the White Rock Ecosystem Restoration Area, Boston Mountain Ranger District of the Ozark-St. Francis National Forest, Arkansas, USA. *k* represents the number of estimated parameters in each model. AIC_c represents the corrected Akaike's Information Criterion for finite sample sizes. ΔAIC_c represents the relative difference in AIC_c to the top model. w_i represents the Akaike model weights. Models with ΔAIC_c <2 were retained along with the full model. (+) Denotes a positive variable relationship to selection probability, (-) denotes a negative relationship, and (.) denotes no significant relationship.

Point Scale				
Model	*k*	AIC_c	ΔAIC_c	w_i
Change in Canopy Cover(+) + Change in Vegetation Height(+)	2	7032.75	0.00	0.34
Canopy Cover(+) + Canopy Cover Heterogeneity(+)	2	7032.79	0.04	0.33
Canopy Cover(+) + Land Cover Type(.)	10	7034.40	1.66	0.15
Global Model	43	7054.72	21.97	0.00

200 M Scale				
Model	*k*	AIC_c	ΔAIC_c	w_i
Canopy Cover(+)	1	7025.03	0.00	0.51
Canopy cover(+) + Canopy Cover Heterogeneity(.)	2	7027.02	1.99	0.19
Canopy Cover(+) + Vegetation Height(.)	2	7027.03	2.00	0.19
Global Model	11	7029.64	4.61	0.05

Table 3 Discrete choice model selection results for nest-site selection by hens from 2012 and 2013 on the White Rock Ecosystem Restoration Area, Boston Mountain Ranger District of the Ozark-St. Francis National Forest, Arkansas, USA. k represents the number of estimated parameters in each model. AIC_c represents the corrected Akaike's Information Criterion for finite sample sizes. ΔAIC_c represents the relative difference in AIC_c to the top model. w_i represents the Akaike model weights. Models with ΔAIC_c <2 were retained along with the full model. (+) Denotes a positive variable relationship to selection probability, (-) denotes a negative relationship, and (.) denotes no significant relationship.

Point Scale				
Model	k	AIC_c	ΔAIC_c	w_i
Canopy Cover Heterogeneity(+)	1	104.9	0.00	0.33
Canopy Cover(.) + Canopy Cover Heterogeneity(+)	2	105.6	0.66	0.24
Canopy Cover(-) + NLCD Cover Type(.)	5	106.5	1.56	0.15
Global Model	13	117.9	13.0	0.00
100 M Scale				
Model	k	AIC_c	ΔAIC_c	w_i
Mean Canopy Cover Heterogeneity(+) + % Area Evergreen Forest(.) + % Area Deciduous Forest(.)	3	100.23	0.00	0.54
Global Model	11	120.26	20.03	0.00
200 M Scale				
Model	k	AIC_c	ΔAIC_c	w_i
Change in Canopy Cover(.) + Change in Vegetation Height(-)	3	102.90	0.00	0.47
Global Model	11	120.07	17.18	0.00

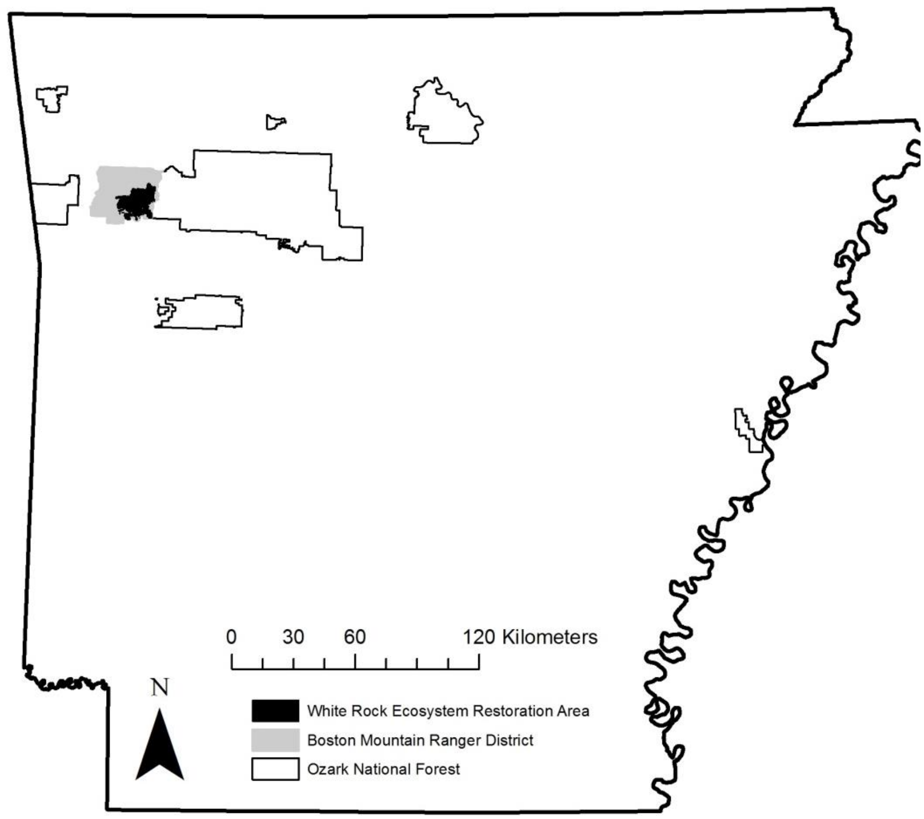

Figure 1. White Rock Ecosystem Restoration Area, Boston Mountain Ranger District of the Ozark-St. Francis National Forest, Arkansas, USA.

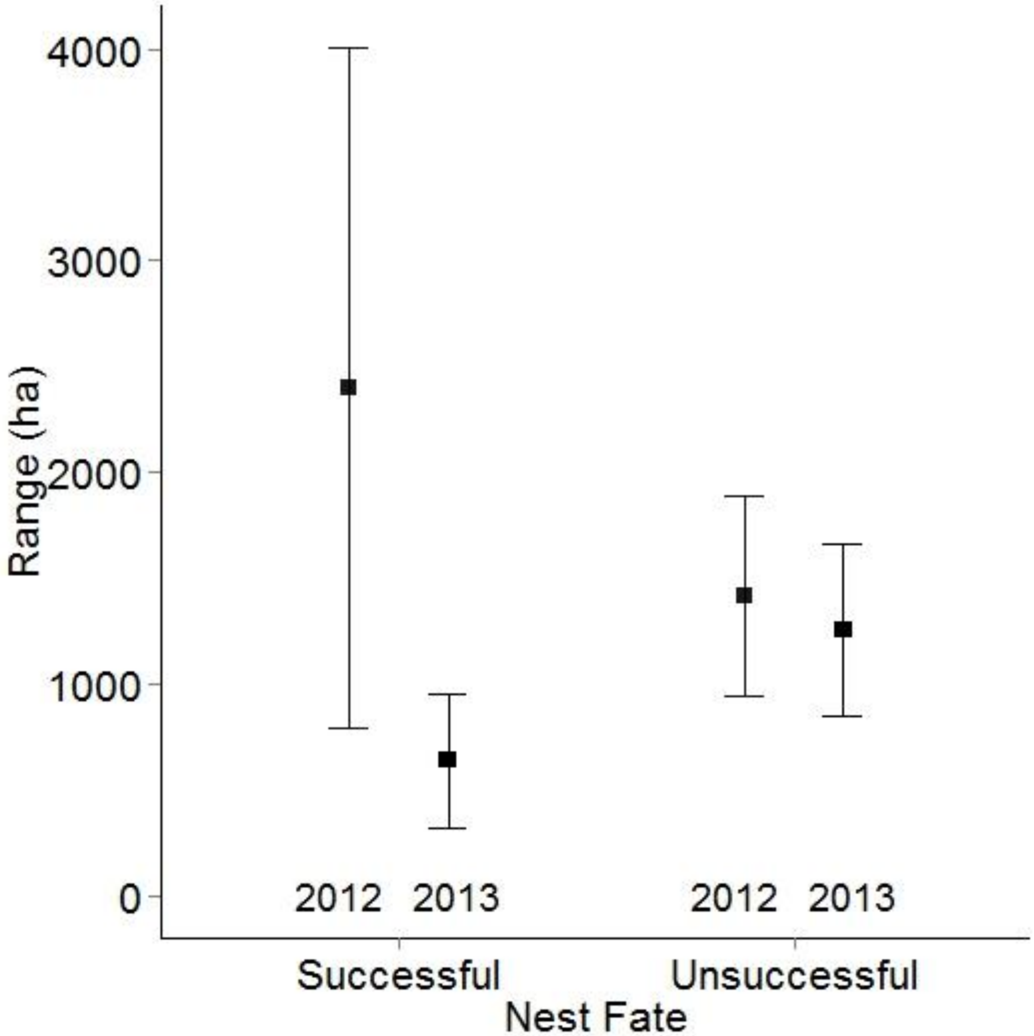

Figure 2. Mean and 95% confidence interval of prenesting ranges (ha) for successful and unsuccessful nesting hens from 2012 and 2013 on the White Rock Ecosystem Restoration Area, Boston Mountain Ranger District of the Ozark-St. Francis National Forest, Arkansas, USA. (A) Represents adult hens and (B) represents juvenile hens.

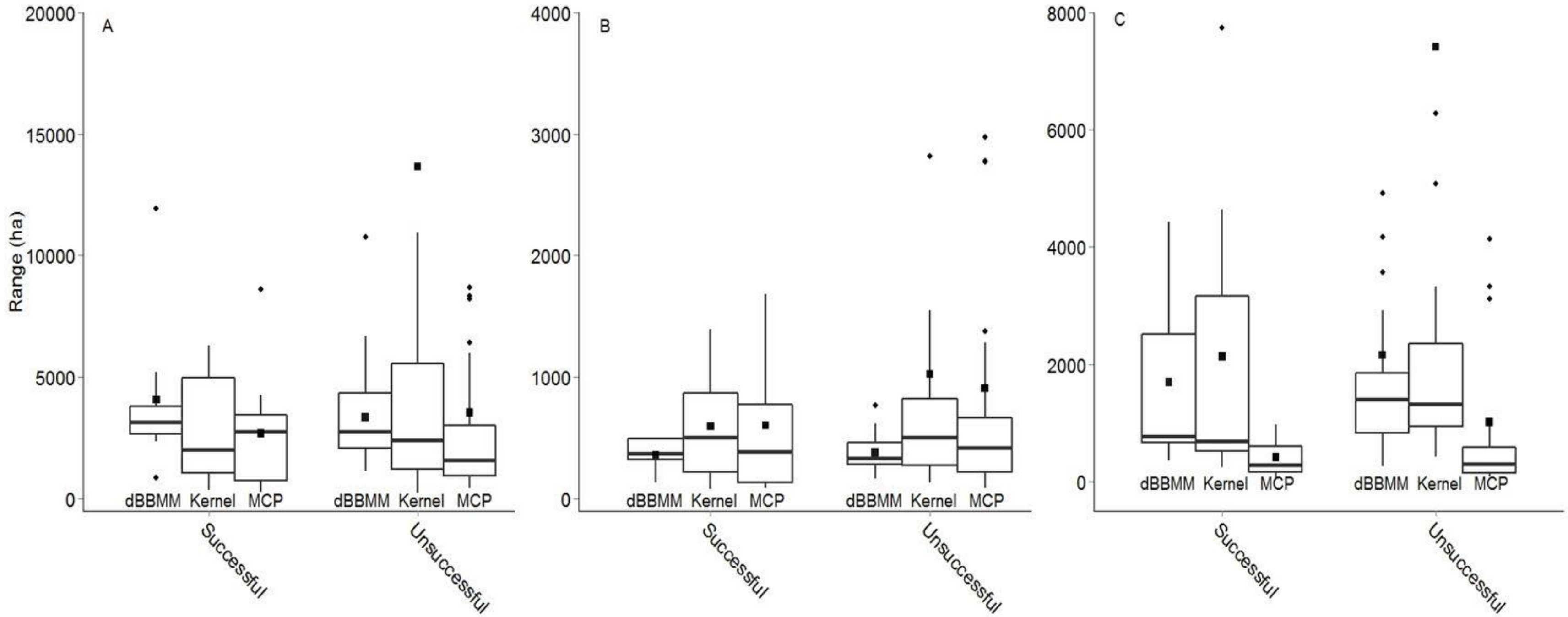

Figure 3. Box and whisker plots of (A) 95% Home Range Kernel, (B) 50% home range core use area, and (C) 95% prenesting range estimates by year for hens from 2012 and 2013 on the White Rock Ecosystem Restoration Area, Boston Mountain Ranger District of the Ozark-St. Francis National Forest, Arkansas, USA. Method used to estimate the home range is annotated in the figure: (dBBMM) dynamic Brownian Bridge Movement Model estimate, (Kernel) kernel density estimate, and (MCP) minimum convex polygon. Black squares (▪) denote the mean range estimates. The mean kernel 95% home range (A) estimate for successful hens is not shown since it was beyond the extent of the figure.

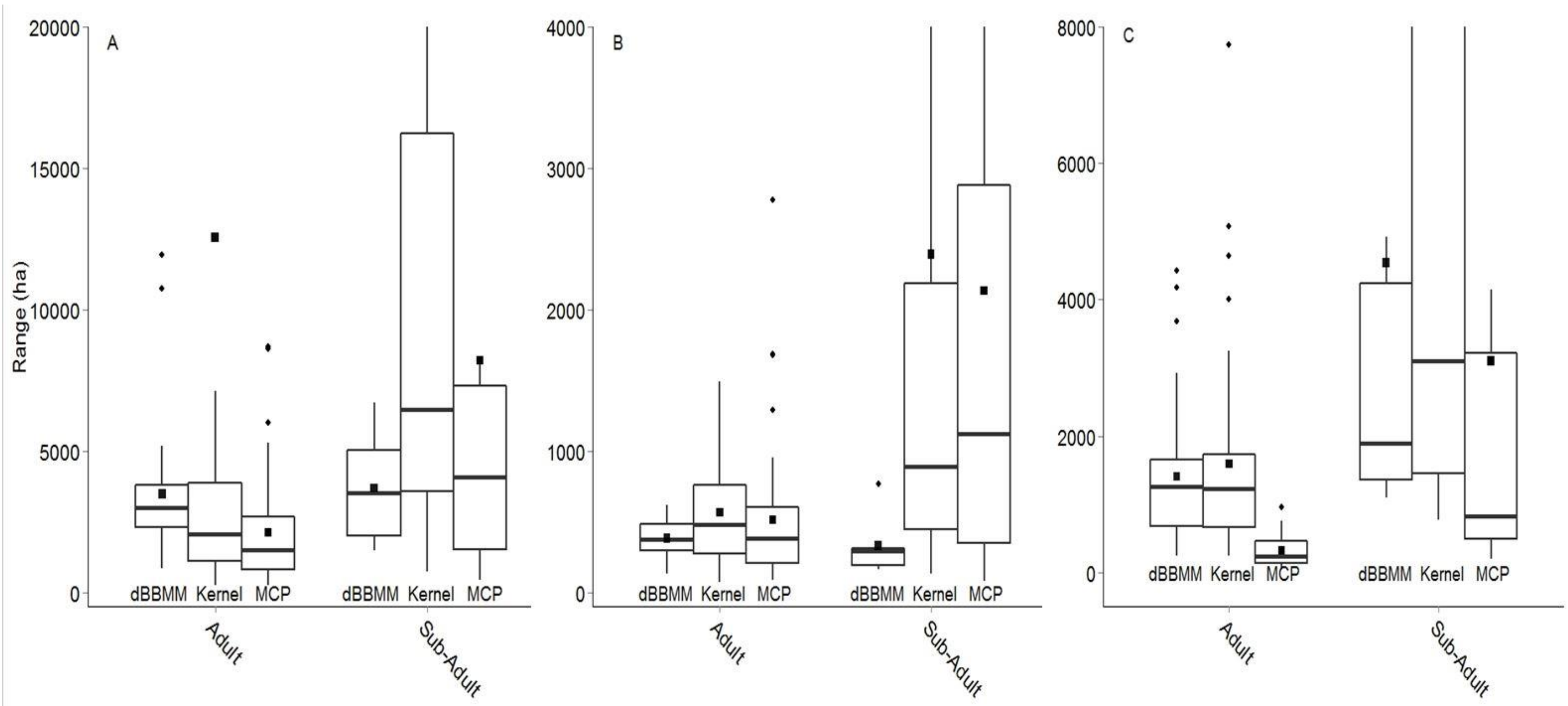

Figure 4. Box and whisker plots of (A) 95% Home Range Kernel, (B) 50% home range core use area, and (C) 95% prenesting range estimates by age class for hens from 2012 and 2013 on the White Rock Ecosystem Restoration Area, Boston Mountain Ranger District of the Ozark-St. Francis National Forest, Arkansas, USA. Method used to estimate the home range is annotated within the figure: (dBBMM) dynamic Brownian Bridge Movement Model estimate, (Kernel) kernel density estimate, and (MCP) minimum convex polygon. Adults have nested at least once while sub-adults have never nested. Black squares (▪) denote the mean range estimates. The mean kernel 95% home range (A) and mean kernel pre-incubation estimate for sub-adults is not shown since it was beyond the extent of the figure.

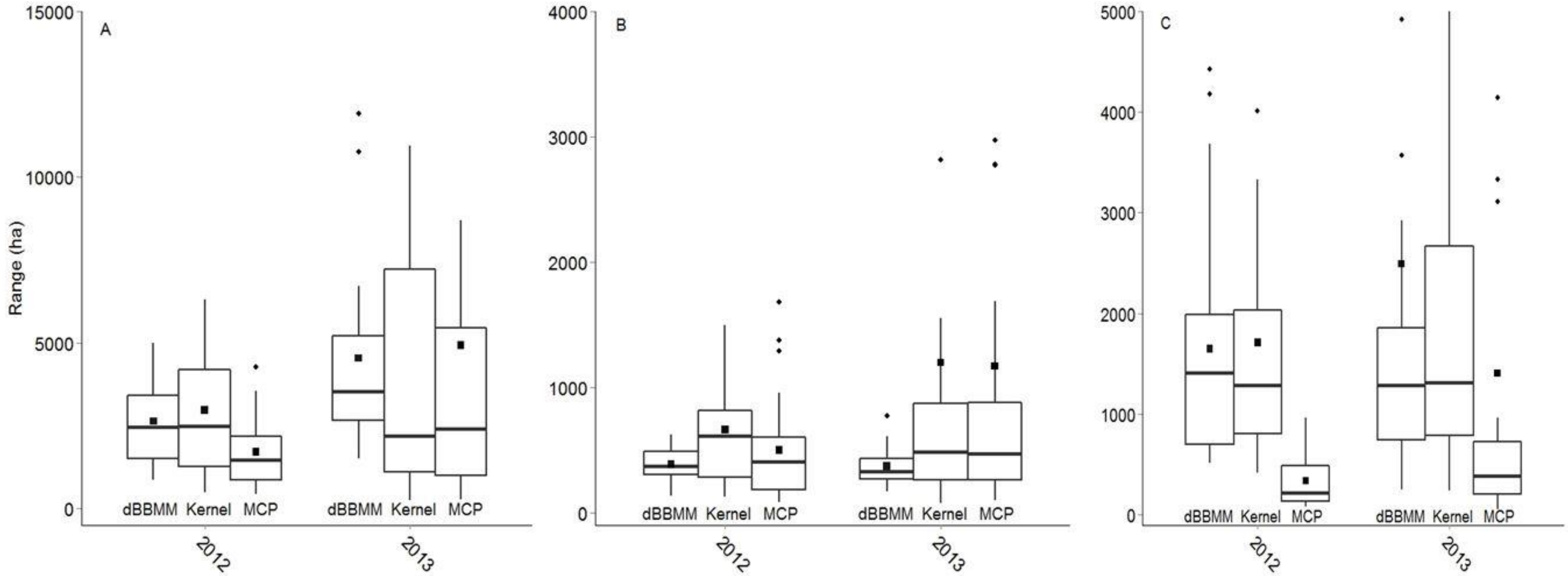

Figure 5. Box and whisker plots of (A) 95% Home Range Kernel, (B) 50% home range core use area, and (C) 95% prenesting range estimates by nest fate for hens from 2012 and 2013 on the White Rock Ecosystem Restoration Area, Boston Mountain Ranger District of the Ozark-St. Francis National Forest, Arkansas, USA. Method used to estimate the home range is annotated within the figure: (dBBMM) dynamic Brownian Bridge Movement Model estimate, (Kernel) kernel density estimate, and (MCP) minimum convex polygon. Black squares (▪) denote the mean range estimates. The mean kernel 95% home range (A) and mean kernel pre-incubation estimate for 2013 is not shown since it was beyond the extent of the figure.

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Appendix A. Discrete choice model selection for the candidate model sets during the pre-incubation period for hens from 2012 and 2013 on the White Rock Ecosystem Restoration Area, Boston Mountain Ranger District of the Ozark-St. Francis National Forest, Arkansas, USA. k represents the number of estimated parameters in each model. AICc represents the corrected Akaike's Information Criterion for finite sample sizes. ΔAICc represents the relative difference in AICc to the top model. w_i represents the Akaike model weights.

Model	k	AIC_c	ΔAIC_c	w_i
VHC + CCC	2	7032.7	0.00	0.34
CC + CCH	2	7032.8	0.04	0.33
CC + LC	10	7034.4	1.66	0.15
LC	9	7035.2	2.49	0.10
CC	1	7036.4	3.70	0.05
CC + VH	2	7038.4	5.70	0.02
VHC + CCC + LC + VHC*LC + CCC*LC	29	7041.7	8.94	0.00
CCH + LC	19	7043.5	10.79	0.00
CCH	1	7045.4	12.63	0.00
LCH	1	7046.2	13.48	0.00
Canopy Cover(CC) + Vegetation Height(VH) + Canopy Cover Heterogeneity(CCH) + Vegetation Height Heterogeneity(VHH) + Canopy Cover Change(CCC) + Vegetation Height Change(VHC) + Land Cover Type(LC) + Land Cover Heterogeneity(LCH) + VHC*LC + CCC*LC	43	7054.7	21.97	0.00
100 m scale				
CC + VH + CCH + VHH + CCC + VHC + LCH + % Evergreen Forest (%EF) + % Deciduous Forest(%DF) + % Pasture/Hay(%PH) + % Grassland/Herbaceous(%GH)	10	7012.9	0.00	0.97
CC + CCH	2	7020.6	7.71	0.02

Appendix A Cont.

Model	k	AIC_c	ΔAIC_c	w_i
CC	1	7023.4	10.51	0.01
CC + VH	2	7024.7	11.77	0.00
VHC + CCC	2	7025.6	12.70	0.00
%DF	1	7029.8	16.89	0.00
%EF + %DF + %PH + %GH	4	7030.5	17.58	0.00
%EF + %DF	2	7031.2	18.30	0.00
CCH + %EF + %DF	3	7033.1	20.25	0.00
%EF + %GH	2	7036.6	23.75	0.00
%PH + %GH	2	7041.1	28.16	0.00
LCH	1	7046.8	33.93	0.00
CCH	1	7046.8	33.95	0.00
200 m scale				
CC	1	7025.0	0.00	0.51
CC + CCH	2	7027.0	1.99	0.19
CC + VH	2	7027.0	2.00	0.19
VHC + CCC	2	7029.1	4.11	0.07
CC + VH + CCH + VHH + CCC + VHC + LCH + %EF + %DF + %PH + %GH	10	7029.6	4.61	0.05
%DF	1	7038.7	13.65	0.00

Appendix A Cont.

Model	k	AIC_c	ΔAIC_c	w_i
CCH + %EF + %DF	3	7039.7	14.72	0.00
%EF + %DF	2	7040.6	15.56	0.00
%EF + %DF + %PH + %GH	4	7042.0	16.97	0.00
CCH	1	7042.4	17.33	0.00
%EF + %GH	2	7043.6	18.53	0.00
%PH + %GH	2	7044.2	19.21	0.00
LCH	1	7047.0	21.93	0.00

Appendix B. Discrete choice model selection for full model sets for nest-site selection by hens from 2012 to 2013 on the White Rock Ecosystem Restoration Area, Boston Mountain Ranger District of the Ozark-St. Francis National Forest, Arkansas, USA. k represents the number of estimated parameters in each model. AICc represents the corrected Akaike's Information Criterion for finite sample sizes. ΔAICc represents the relative difference in AICc to the top model. wi represents the Akaike model weights.

Model	k	AIC_c	ΔAIC_c	w_i
CCH	1	104.9	0.00	0.33
CC + CCH	2	105.6	0.66	0.24
CC + LC	5	106.5	1.56	0.15
CC	1	107.3	2.37	0.10
LC	4	108.2	3.27	0.06
LCH	1	108.7	3.75	0.05
CC + VH	2	109.3	4.39	0.04
CCC + VHC	2	110.2	5.34	0.02
Canopy Cover(CC) + Vegetation Height(VH) + Canopy Cover Heterogeneity(CCH) + Canopy Cover Change(CCC) + Vegetation Height Change(VHC) + Land Cover Type(LC) + Land Cover Heterogeneity(LCH)	10	116.6	11.68	0.00
100 m scale				
CCH + %EF + %DF	3	100.23	0.00	0.54
CCH	1	102.98	2.75	0.14
%DF	1	103.91	3.69	0.09
%EF + %GH	2	104.42	4.20	0.07

Appendix B Cont.

Model	k	AIC_c	ΔAIC_c	w_i
CCC + VHC	2	104.93	4.70	0.05
CC + CCH	2	105.14	4.92	0.05
%EF + %DF	2	105.17	4.94	0.05
%EF + %DF + %PH + %GH	4	108.33	8.11	0.01
CC	1	109.03	8.80	0.01
LCH	1	109.68	9.45	0.00
%PH + %GH	2	109.90	9.68	0.00
CC + VH	2	111.20	10.97	0.00
CC + VH + CCH + CCC + VHC + LCH + % Evergreen Forest (%EF) + % Deciduous Forest(%DF) + % Pasture/Hay(%PH) + % Grassland/Herbaceous(%GH)	9	115.53	15.31	0.00
200 m scale				
CCC + VHC	2	102.9	0.00	0.46
CCH + %EF + %DF	3	105.0	2.15	0.16
CCH	1	105.9	3.01	0.10
%EF + %GH	2	106.6	3.72	0.07
%DF	1	106.8	3.90	0.07
CC + CCH	2	108.1	5.19	0.03
%EF + %DF	2	108.2	5.28	0.03

Appendix B Cont.

Model	k	AIC_c	ΔAIC_c	w_i
LCH	1	108.6	5.75	0.03
CC	1	109.4	6.53	0.02
%PH + %GH	2	110.3	7.40	0.01
%EF + %DF + %PH + %GH	4	111.1	8.24	0.01
CC + VH	2	111.6	8.69	0.01
CC + VH + CCH + CCC + VHC + LCH + %EF + %DF + %PH + %GH	9	115.4	12.53	0.00

Appendix C. Variable names and definitions

Variable	Definition
Canopy Cover(CC)	Percent canopy cover (0-100%)
Vegetation Height(VH)	Vegetation height in meters
Canopy Cover Heterogeneity(CCH)	Value representing the number of different values of canopy cover surrounding the point
Vegetation Height Heterogeneity(VHH)	Value representing the number of different values of vegetation height surrounding the point
Canopy Cover Change(CCC)	Change in percent canopy cover from 2001 to 2010 (%)
Vegetation Height Change(VHC)	Change in vegetation height from 2001 to 2010 (m)
Land Cover Type(LC)	National Land Cover Dataset (NLCD) cover type at that point
Land Cover Heterogeneity(LCH)	Value representing the number of different NLCD cover types surrounding the point
% Evergreen Forest (%EF)	Percent evergreen forest - area dominated by trees generally greater than 5 meters tall, and greater than 20% of total vegetation cover. More than 75% of the tree species maintain their leaves all year. Canopy is never without green foliage (Homer et al 2007).
% Deciduous Forest(%DF)	Percent deciduous forest - area dominated by trees generally greater than 5 meters tall, and greater than 20% of total vegetation cover. More than 75% of the tree species shed foliage simultaneously in response to seasonal change (Homer et al 2007).
% Pasture/Hay(%PH)	Percent pasture/hay – area of grasses, legumes, or grass-legume mixtures planted for livestock grazing or the production of seed or hay crops, typically on a perennial cycle. Pasture/hay vegetation accounts for greater than 20% of total vegetation (Homer et al 2007).
% Grassland/Herbaceous(%GH)	Percent grassland/herbaceous - area dominated by graminoid or herbaceous vegetation, generally greater than 80% of total vegetation. These areas are not subject to intensive management such as tilling, but can be utilized for grazing (Homer et al 2007).

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MEMORANDUM

TO: David Krementz

FROM: Craig N. Coon, Chairman
Institutional Animal Care
And Use Committee

DATE: November 9, 2010

SUBJECT: IACUC PROTOCOL **APPROVAL**
Expiration date : **November 8, 2013**

The Institutional Animal Care and Use Committee (IACUC) has **APPROVED** Protocol **#11012- "EFFECTS OF LARGE SCALE GROWING SEASON PRESCRIBED BURNS ON MOVEMENTS, HABITAT USE, PRODUCTIVITY, AND SURVIVAL OF FEMALE WILD TURKEYS ON THE WHITE ROCK ECOSYSTEM RESTORATION PROJECT OF THE OZARK-ST. FRANCIS NATIONAL FOREST"**. You may begin this study immediately.

In granting its approval, the IACUC has approved only the protocol provided. Should there be any changes in the protocol during the research, please notify the IACUC in writing **prior** to initiating the changes. If the study period is expected to extend beyond, **11-08-2013** you must submit a new protocol. By policy the IACUC cannot approve a study for more than 3 years at a time.

The IACUC appreciates your cooperation in complying with University and Federal guidelines for research involving animal subjects.

cnc/car

cc: Animal Welfare Veterinarian

UNIVERSITY OF ARKANSAS

J. William Fulbright College of Arts and Sciences
Department of Biological Sciences

Chapter 2, "Impacts of Landscape Level Woodland and Savanna Restoration on Eastern Wild Turkey Movement and Reproductive Ecology" of H. T. Pittman's dissertation is intended for submission for publication with one coauthor, D. G. Krementz.

I, Dr. David G. Krementz, advisor of Henry Tyler Pittman, confirm Henry Tyler Pittman will be first author and completed at least 51% of the work for this manuscript.

David G. Krementz Unit Leader U. S. Geological Survey Arkansas Cooperative Fish and Wildlife Research Unit	Date

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Chapter III

Efficacy of landscape scale woodland and savanna restoration at multiple spatial and temporal scales

Henry Tyler Pittman

And

Dr. David G. Krementz

Abstract

The loss of woodland and savanna ecosystems in the Central Hardwoods region has caused managers to begin managing forests to restore woodland and savanna characteristics to their historical ranges. Over the past two decades forest managers have implemented woodland and savanna restoration at the landscape level (≥10,000 ha) in the Central Hardwoods, specifically early growing season prescribed fire. We initiated our study to examine the impact and efficacy of early growing season prescribed fire as restoration tool on vegetation characteristics. We collected vegetation measurements at 70 locations within and around the White Rock Ecosystem Restoration Area (WRERA), Ozark-St. Francis National Forest, Arkansas, USA. We used generalized linear models to understand how different management scenarios impacted vegetation structure and composition. We found the number of large shrubs was negatively related and small shrubs were positively related to prescribed fire severity. We also found that visual concealment from ground level to 1 m in height was positively related to time since prescribed fire and woody ground cover was negatively related to the number of prescribed fire treatments. We also used LANDFIRE datasets to summarize and assess the efficacy of the use of prescribed fire only for restoration on a landscape scale. We found that since the initiation of early growing season prescribed fire canopy cover had been reduced but not to levels characteristic of woodlands and savannas. As a management practice early growing season prescribed fire alone was not effective at restoring woodland and savanna conditions after 8 years on the WRERA.

Introduction

Oak woodlands and savannas historically covered large extents of central North America, creating a transition zone between closed canopy deciduous forest in the east and tall grass prairie in the Central Plains (Dyksterhuis 1957; Anderson 1983; Nuzzo 1985). Woodland and savanna ecosystems consisted of open canopies and diverse understories of grasses, forbs, and some woody shrubs (McPherson 1997; Taft 1997; Anderson et al. 1999). These ecosystems were maintained by disturbance, typically fire that prevented canopy closure, reduced shrub competition, and promoted the presence and persistence of fire adapted species. After European settlement, conversion to agriculture and fire suppression significantly reduced the extent of oak woodlands and savannas. Nuzzo (1985) estimated that less than 1% of oak woodland and savanna ecosystems still exist in their historical conditions.

During the past 30 years managers realized that the loss of woodland and savanna ecosystems affected plant and animal species that require the early successional and transitional characteristics of these systems. Managers are attempting to restore former woodland and savanna sites to their historic structure and composition to create wildlife habitat, protect fire adapted species, increase advance oak regeneration, and eliminate oak competitors (Sparks et al. 1998; Peterson and Reich 2001; Hutchinson et al. 2005). Restoration has mostly targeted sites currently in closed canopy forest with one or some combination of mechanical canopy removals, herbicide treatments, and prescribed fire (Jackson and Buckley 2004). In combination with restoration efforts numerous research studies have attempted to better understand the role disturbance has in these systems and the efficacy of restoration techniques.

Research has examined the techniques, timing, and results of restoration in many upland hardwood ecosystems in North America. Many of these studies have addressed the reintroduction of fire, season of fire, intensity of fire and the effectiveness of fire as a system maintaining disturbance (Sparks et al. 1998; Peterson and Reich 2001; Hutchinson et al. 2005; Franklin et al. 2003). Prescribed fire has been studied in combination with both mechanical treatments and herbicide treatments (Albrecht and McCarthy 2006; Brose et al. 1999; Jackson and Buckley 2004). A majority of prescribed fire and restoration research in upland hardwood ecosystems has focused on its ability to reduce competition by weedy species and increase advanced oak regeneration (Blankenship and Arthur 2006; Hutchinson et al. 2005; Iverson et al. 2008). Some studies have also addressed the conversion of closed canopy hardwood forest back to woodland and savanna conditions using various techniques in combination with prescribed fire (Hutchinson et al. 2005; Albrecht and McCarthy 2006; Brose et al. 1999). Researchers have also studied the impacts of restoration on wildlife and its creation or maintenance of habitat many wildlife species require (Brawn et al. 2001; Davis et al. 2000; Teidemann et al. 2000; Reidy et al. 2014). Nuzzo et al. (1996), Sparks et al. (1998), and others have examined the understory vegetation responses to prescribed fire and oak woodland and savanna management. However, many of these research studies have addressed restoration and it components in experimental settings and at relatively small scales (≤500 ha) compared to current landscape level implementation.

In this study we will examine restoration management on a landscape level (≥10,000 ha) as it is currently implemented. We plan to (1) describe vegetation structural changes over multiple spatial and temporal scales as a result of restoration treatments and (2) describe the efficacy of current restoration techniques at a landscape level. We will describe vegetation characteristics that will have meaning for wildlife habitat and forest production. We plan to use both ground collected micro-vegetation scale data in combination with landscape scale vegetation data derived from various ground collected and remotely sensed sources. Our analysis will cover immediate vegetation responses (≤6yrs) to restoration treatments and long term responses (>6 yrs) to repeated treatments. Our research will be essential in shaping future management decision about scale and technique throughout the Central Hardwoods region. Our work will also provide knowledge on upland hardwood management for wildlife and forest product managers that can be applied in the Central Hardwoods region and other upland hardwood ecosystems in North America.

Methods

Study Area

The White Rock Ecosystem Restoration Area (WRERA) consists of 16,380 ha of upland hardwood and pine ecosystems in Northwest Arkansas, United States of America (USA; Fig 1). It is part of the larger main division of the Boston Mountain Ranger District (41,400 ha) on the Ozark-St. Francis National Forest. The WRERA is a high priority woodland and savanna restoration area for the United States Department of Agriculture (USDA) Forest Service. Historically, the WRERA was dominated by woodlands and savannas maintained by frequent fire disturbances (Cutter and Guyette 1994; Batek et al. 1999; Chapman et al. 2006; Guyette et al. 2006; Foti 2004; Stambaugh and Guyette 2006). After fire suppression during the 20th century, the WRERA became dominated by closed canopy hardwood forests of various oak (*Quercus sp.*) and hickory (*Carya sp*.) species. Understories consisted of canopy species regeneration, Marshall black gum (*Nyssa sylvatica*), flowering dogwood (*Cornus florida*), Carolina buckthorn (*Frangula caroliniana*), black berry (*Rubus spp.*) and devil's walking-stick (*Aralia spinosa*; USDA, NRCS 2014). Pine ecosystem canopies were dominated by short leaf pine (*Pinus echinata*) while understories consisted of hardwood and pine regeneration. Since 2002, the WRERA has been intensively managed with large scale early growing season prescribed fire to restore historic woodland and savanna conditions. Management prescription for WRERA describes woodlands as having "open canopies, sparse mid-stories and well-developed understories that are typically dominated by grasses and forbs, but also may become shrubby between fires and have a significant woody component" with 40-60% canopy closure (Ozark NF Plan 2005). The prescription also describes management techniques available to achieve these objectives including mechanical canopy removal, herbicide, and fire treatments. However, mechanical and herbicide treatments have been mostly absent in the restoration area since 2002. The WRERA is divided into 16 prescribed fire units ranging from 467 to 1,670 ha in size. Early growing season prescribed fire has been implemented on a three to five yr rotation in 15 of the 16 prescribed fire units, and units received one to four prescribe fire treatments since 2002.

Vegetation Data Collection

We collected vegetation measurements at 70 locations during June and July from 2011 to 2013. We stratified locations by time since prescribed fire and cover type, and sampled each location once a year. We also collected vegetation measurements at reference sites with no history of fire according to the impact/reference design of van Mantgem (2001). Vegetation measurements included visual concealment from zero to one meter in height (Nudds 1977), percent ground cover type (Daubenmire 1959), tree counts (diameter classes at breast height (dbh)), shrub counts (diameter classes at ground level), understory height (m), canopy cover (Lemmon 1956), and a fire severity index (Cocking et al. 2014). The small shrub size category included advanced oak regeneration, a common measurement collected in studies examining the response of upland hardwood ecosystems to prescribed fire.

In landscape scale analyses, we used 2001 and 2010 LANDFIRE vegetation datasets (Rollins 2009) for canopy cover and vegetation height and 2001 and 2011 National Land Cover Datasets (NLCD) for forest cover type (Homer et al. 2007; Jin et al 2013). We used the raster calculator and focal statistics tools in ArcGIS to develop datasets for percent change in canopy cover from 2001 to 2010, canopy cover heterogeneity for 2001 and 2010, vegetation height heterogeneity for 2001 and 2010, and NLCD cover type heterogeneity for 2001 and 2011 (ESRI 2013). We incorporated all datasets into a geographic information system (GIS) with existing USDA Forest Service GIS data.

Data Analysis

We used means and 95% confidence intervals for each ground collected vegetation variable to determine if any trends existed or year effects consistent with the impact/reference design (van Mangtem 2001). We fitted generalized linear models (GLM) to each vegetation variable to determine what management factors influenced vegetation trends. Model responses were vegetation variables and predictors were variables related to year, fire severity, and management practices. All GLMs were fitted using a normal distribution. Each model set consisted of 11 candidate models of predictors including time since prescribed fire (yrs), year the sample was collected (2011-2013), a plot fire severity index (0-1), and the number of prescribed fire treatments (1-3). We also included terms for plot fire severity and time since prescribed fire interactions with the year the sample was collected. We used Akaike's Information Criterion corrected for finite sample size (AIC_c) to rank candidate models and model averaged parameter estimates of the top model ($\Delta AIC_c \leq 2$) to account for any model selection uncertainty (Burnham and Anderson 2002). We only report model averaged parameter estimates for vegetation variable model sets that produced one or more candidate model with $\Delta AIC_c \geq 2$ of the best performing intercept only or the global model. We performed all statistical modeling and analyses using R statistical language (R Core Development Team, 2014).

We summarized all 14 derived vegetation datasets for each of the 16 prescribed burn units. Using these summaries we calculated the proportional area of each burn unit over the range of possible variable value. We plotted these proportional areas for all prescribed burn units in bar plots to compare the distributions of area based on the number of prescribed fire treatments (package "ggplot2"; R Core Development Team, 2014). We visually examined these plots for shift in distributions explained by the number of prescribed fire treatments.

Results

Micro Vegetation Scale

Our results show the most vegetation variability occurred the same year units were burned. We found the greatest annual mean fluctuations occurred 1 to 3 and 4 to 6 years after treatment. Mean variable values indicated that prescribed fire units treated 1-6 years before sampling had higher visual concealment (0-1m), understory vegetation height, and number of small shrubs than unburned units (Table 1). Confidence intervals indicated that prescribed fire units treated 1-6 years before sampling had more annual variability in understory height and the number of small shrubs. We found woody vegetation dominated ground cover in all time since fire categories and years. We observed units untreated or treated within the past year had higher percent woody ground cover in 2011, but dropped sharply in 2012 and did not recover (Table 1).

We also observed overall low levels of grass and forb cover in all time since fire categories and years. Forb cover decreased in 4-6 yrs since fire and untreated units in 2012. Forb cover was also highly variable in all time since fire categories and years. We did not observe an expected increase in grass and forb cover immediately following prescribed fire in 0 yrs since fire units.

Model averaged parameter estimates from generalized linear models indicated visual concealment, understory height, and the number of small shrubs increased as fire severity increased (Table 2). We observed a negative relationship between fire severity and the number of medium trees, large trees, and large shrubs. We found a positive year effect on severity existed for 2013 for large shrubs and a negative year effect on severity for small shrubs in 2012. We also found a negative year effect on time since fire in 2013 on visual concealment (0-1 m), a positive year effect on time since fire for the number of medium trees and a negative effect of the number of prescribed fire treatments on woody ground cover.

Landscape scale

We examined the distributions of percent change in canopy cover for 16 prescribed fire units and found in all units treated with prescribed fire that there was a negative percent change in canopy cover from 2001 to 2010 (Fig 2) There was a positive percent change in canopy cover from 2001 to 2010 in the untreated unit (Fig 2). We found the percent change in canopy cover from 2001 to 2010 in units treated with prescribed fire was negative and it only reduced the majority of the units' areas to 75% canopy cover (Fig 3). We found in the 15 units treated with fire, approximately 5% (~1,000 ha) were less than or equal to the 60% canopy cover criteria listed in management prescriptions for woodlands. Before woodland restoration treatments, we observed canopy cover heterogeneity scores for all units centered near a score of 2 (Fig 4). After application of prescribed fire treatments we observed a shift in the heterogeneity distributions of all units towards one, more homogeneous habitat (Fig 5). We observed no major differences in canopy cover response after one, two, or three prescribed fire treatments to levels consistent with woodlands and savannas.

Discussion

We found that both percent visual concealment (0-1 m), and understory height was greater in units treated with prescribed fire compared to untreated units. However, levels immediately following prescribed fire did not increase beyond untreated levels, indicating the response of understory vegetation to prescribed fire may not be realized until 1 to 3 years after a fire treatment. Increased levels of vegetation variables following fire persisted 4 to 6 years after prescribed fire treatment. We observed higher levels of forb ground cover in treated units compared to untreated units, but did not observe the same delayed response after fire. The delayed response in forest structure variables, but not in composition variables, suggest the response of forest structure following fire treatment required more time and multiple growing seasons to replace the above ground biomass lost during prescribed fire treatments. Hmielowski (2013) observed short-term fire effects in hardwood ecosystems and found the response and magnitude of response in vegetation is influenced by the size of root reserves of plants at the time of top-kill. We observed delayed increases in the number of small shrubs consistent with Hmielowski's observations. These small shrubs contribute significantly to forest structure and mostly result from sprouting root reserves of top-killed species. The short term response of forest structure to prescribed fire is an important ecosystem characteristic, because it may affect wildlife species that require certain forest structure characteristics (Reidy 2014; Badyaev 1994).

We found that vegetation structure was influenced by fire severity. The presence of the fire severity variable in our top models suggested that structural responses were influenced by the intensity of fire consistent with other prescribed fire studies (Arthur et al. 1998; Elliott and Vose 2005; Brose et al. 1999; Hutchinson et al. 2005; Peterson and Reich 2001). Fire severity negatively impacted woody ground cover, large shrubs, and medium trees similar to the reduction of mid-story species found by others (Nuzzo et al. 1996; Arthur et al. 1998). We observed positive effects of fire severity on structural variables and small shrubs. This positive relationship suggest more intense prescribed fires have increased advanced oak regeneration in these upland hardwood ecosystems, similar to results found in studies of management focused on forest products (Arthur et al. 1998; Brose et al. 1998; Albrect and McCarthy 2005; Iverson et al. 2007). The increase in advance oak regeneration in these stands will benefit forest health by eventually replacing declining canopy trees, a management concern in the Central Hardwoods region. However, with increased advanced oak regeneration and limited canopy opening, woodland and savanna structure and composition will be difficult to achieve.

Rapid changes of forest composition in response to prescribed fire treatments have been documented in hardwood ecosystems in the Central Hardwoods region. We did not find the same magnitude of responses in herbaceous, grass and forb, ground cover (10-15%) that has been observed in other woodland and savanna ecosystems (approximately 30%; Hartman and Heumann 2003; Hutchinson et al. 2005; Nuzzo et al. 1996; Taft 2003). We also did not observe increases in grass ground cover immediately following prescribe fire, and in subsequent years as have other studies of woodland and savanna vegetation responses to prescribed fire (Hutchinson et al. 2005; Sparks et al. 1998). Franklin et al. (2003) found that vegetation responses after prescribed fire treatments were dependent on the extant forest composition. Since there was an absence of a significant herbaceous component in the under story at our study site, due to fire suppression, Franklin et al.'s (2003) conclusion could be one explanation for the limited response of herbaceous ground cover. Another potential explanation could be that prescribed fire treatments alone do not allow enough sunlight to the forest floor to initiate a response by grasses. Other studies have found that more severe disturbances, such as mechanical canopy removals or more frequent fire, are necessary to obtain significant responses and changes in the understory compositions (Franklin et al. 2003; Peterson and Reich 2001; Elliott and Vose 2005; Hutchinson et al. 2005). Our observation of increased small shrub counts and no major changes in canopy cover at the landscape level support the idea that light reaching the understory is limiting herbaceous vegetation responses.

In 2012, we observed a decrease in percent visual concealment, woody, and herbaceous ground cover which may have resulted from an extreme drought that occurred that year (index – D4 out of 5, U.S. Drought Monitor 2014). Other studies have suggested that vegetation response differs over a moisture gradient but few have documented the impact of severe drought on the vegetation response to prescribed fire treatments (Harmon et al. 1984; Hollingsworth et al. 2013; Anning et al. 2014). We observed the most severe drought effects immediately following prescribed fire treatments and in unburned units compared to units treated one to six years before sampling, where the effects of drought were less severe. Drought effects were also indicated by our model parameter estimates where year predicted multiple vegetation responses (Table 2). Our model results indicated a delayed effect of drought resulting from increased fire severity that reduced medium trees and large shrubs. Large shrubs and medium trees do not contain the amount of root reserves as large canopy trees and likely entered the 2013 prescribed fire season in a drought stressed state as opposed to other years. These observed drought effects are one possible explanation for the site-specific nature and high variability among prescribed fire studies.

We also examined the large spatial and temporal scales of woodland and savanna restoration, specifically the effects of the use of prescribed fire. Our results suggest prescribed fire only is not restoring woodlands and savannas on a landscape scale after 8 yrs. If prescribed fire alone had been effective, we would expect to see changes in canopy cover similar to those created by mechanical canopy reductions or natural reductions in canopy like insect or ice damage (Fig 6). Our finding of no change in canopy cover were consistent with other restoration studies that found the combination of mechanical canopy removals and frequent fire were necessary to produce conditions most similar to woodlands and savannas (Franklin et al. 2003; Hutchinson et al. 2005; Elliott and Vose 2005; Dey and Hartman 2005). Although fire alone over the past 8 years has proven less effective at restoring and maintaining woodlands and savannas on the WRERA, that treatment may be effective over a longer temporal scale, potentially 50 to 100 years (Baker 1994; Hartman and Heumann 2003).

Our analysis of canopy cover heterogeneity for WRERA indicated that the landscape had become more homogenous since the implementation of restoration. If prescribed fire alone were having the desired restoration effects, we would have expected the opposite of the observed outcome. The decrease in heterogeneity is likely due to a lack of canopy reductions from mechanical efforts and low intensity prescribed fires. Prescribed fires are implemented on the WRERA using aerial ignition, a method that has been found to be less intense than a typical head fire or a natural fire and results in few if any canopy openings (Johansen 1987; Price et al. 2007). However, the effects of aerial ignition techniques on landscape patch dynamics are under studied and likely site specific.

Our findings of only limited changes in forest structure after 8 years of periodic prescribed fire (~3 to 5 year rotation) were consistent with the hypothesized (20+ years) duration of prescribed fire treatment by Hartman and Heumann (2003) in Missouri Ozarks necessary to restore woodlands and savannas. We can also expect the needed application of periodic prescribed fire to be even longer, possibly 75 to 150 years. Baker (1994) in a simulation study of the restoration of forest structure after fire suppression in northern Minnesota indicated that landscape structure could be restored under a natural fire regime after 50 to 75 years. However, simulations using the LANDIS model in the Missouri Ozarks to determine the future impact of mechanical disturbance on the landscape characteristics, suggests that even under the most intensive mechanical harvest regime likely to be implemented on public lands, changes in forest structure could take 75 to 120 years to achieve (Shifley et al. 2006). The WRERA with only the use of early growing season prescribed fire is likely most similar to the moderate mechanical management scenarios simulated by Shifley et al. (2006) that required greater than 100 years to produce overall shifts in forest structure. Admittedly these simulations were based on timber harvest, but an periodic early growing season prescribed fire regime and its impact on tree mortality could be considered analogous to moderate or low intensity uneven-aged harvest scenarios. Changes to forest structure in the Ozarks could also be slowed even further by variations in characteristics of sites being managed for woodland and savanna restoration. According to Foti (2004) high variability in vegetation responses should be expected due to the underlying geological substrate in the Boston Mountains and the Ozark Highlands. Taking into account all of these factors restoration of the woodland and savanna structure to the WRERA is likely to take from 25 years on ideal sites to upwards of 100 years on sites less suitable for restoration.

Conclusions

We found the use of prescribed fire only for landscape level restoration of woodlands and savannas in the WRERA has not achieved woodland and savanna characteristics over large areas. Our findings indicate that prescribe fire has changed vegetation structure and composition, but in some instances those changes have not been in the intended direction or to the intended magnitude. Prescribed fire has increased advanced oak regeneration, one of the management objectives of the USDA Forest Service. However, without significant canopy reduction advanced regeneration will be limited in success, and woodland and savanna conditions will not be achieved in the foreseeable future or to the extent desired. Continued monitoring of forest conditions will be necessary to determine if management activities begin to create woodland and savanna conditions.

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Table 1. Mean variable values and 95% confidence intervals for time since prescribe fire from 2011 to 2013.

Variable	Time Since Prescribed Fire	Year 2011	Year 2012	Year 2013
Percent visual concealment (0-1m)	0 yrs Since Burn	57.27 (40.54,74.01)	52.35 (35.17,69.54)	58.33 (32.82,83.85)
	1-3 yrs Since Burn	71.46 (61.63,81.30)	63.45 (50.50,76.40)	75.25 (66.10,84.40)
	4-6 yrs Since Burn	63.85 (50.80,76.89)	71.76 (58.61,84.92)	51.58 (39.29,63.87)
	No Burn Record	60.00 (46.76,73.24)	40.00 (19.76,60.24)	28.00 (-1.33,57.33)
Understory height (m)	0 yrs Since Burn	0.85 (0.59,1.12)	0.68 (0.41,0.94)	0.85 (0.52,1.18)
	1-3 yrs Since Burn	1.17 (0.98,1.36)	1.10 (0.85,1.36)	1.27 (1.08,1.46)
	4-6 yrs Since Burn	1.08 (0.82,1.33)	1.31 (1.03,1.58)	0.80 (0.60,1.00)
	No Burn Record	0.87 (0.63,1.11)	0.58 (0.24,0.93)	0.68 (0.12,1.24)
No. small shrubs (≤4 cm)	0 yrs Since Burn	13 (6,19)	11 (5,17)	10 (1,18)
	1-3 yrs Since Burn	30 (22,38)	23 (18,29)	31 (23,39)
	4-6 yrs Since Burn	25 (18,32)	31 (22,41)	13 (9,17)
	No Burn Record	18 (12,24)	18 (8,28)	15 (4,25)
Percent woody ground cover	0 yrs Since Burn	40.45 (23.25,57.66)	7.94 (0.19,15.69)	6.67 (-4.55,17.89)
	1-3 yrs Since Burn	13.90 (6.99,20.82)	17.41 (7.59,27.23)	14.25 (6.30,22.20)
	4-6 yrs Since Burn	4.23 (-0.96,9.42)	7.06 (-2.23,16.34)	14.47 (3.85,25.10)
	No Burn Record	27.89 (15.13,40.66)	3.33 (-1.51,8.18)	8.00 (-1.09,17.09)

Table 1 Cont.			Year	
Variable	Time Since Prescribed Fire	2011	2012	2013
Percent grass ground cover	0 yrs Since Burn	0 (0,0)	0 (0,1)	0 (0,0)
	1-3 yrs Since Burn	2 (0,4)	4 (-1,8)	4 (1,7)
	4-6 yrs Since Burn	9 (1,17)	0 (0,0)	3 (0,7)
	No Burn Record	2 (0,3)	2 (0,4)	0 (0,0)
Percent forb ground cover	0 yrs Since Burn	11.48 (1.17,21.79)	9.64 (2.06,17.22)	17.08 (0.23,33.94)
	1-3 yrs Since Burn	12.78 (6.39,19.17)	11.56 (2.62,20.49)	13.60 (6.23,20.96)
	4-6 yrs Since Burn	12.31 (2.71,21.91)	7.65 (0.31,14.99)	2.63 (0.31,4.96)
	No Burn Record	4.34 (-4.17,12.85)	0.42 (-0.40,1.23)	0.00 (0.00,0.00)

Table 2. List of model averaged parameter estimates (≤ 2 ΔAIC) from generalized linear models for each vegetation structure variable.

	β	Lower 95% CI	Upper 95% CI
Visual concealment (0-1 m)			
Fire Severity	0.369	0.244	0.494
Time Since Fire:Year 2013	-0.161	-0.315	-0.008
Understory Height			
Fire Severity	0.369	0.243	0.495
Canopy Cover			
Fire Severity:Year2012	22.76	14.56	30.96
Woody Stem			
Number of Prescribed Fires	-0.141	-0.275	-0.008
Year 2012	-0.149	-0.297	-0.002
Medium Trees			
Fire Severity	-0.244	-0.375	-0.114
Time Since Fire:Year 2013	0.237	0.077	0.397
Large Trees			
Fire Severity	-0.38	-0.551	-0.209
Small Shrubs			
Fire Severity	0.415	0.161	0.669
Fire Severity:Year 2012	-0.223	-0.394	-0.051
Large Shrubs			
Year2013	-0.294	-0.469	-0.12
Fire Severity	0.233	0.004	0.462
Fire Severity:Year2013	0.502	0.237	0.767

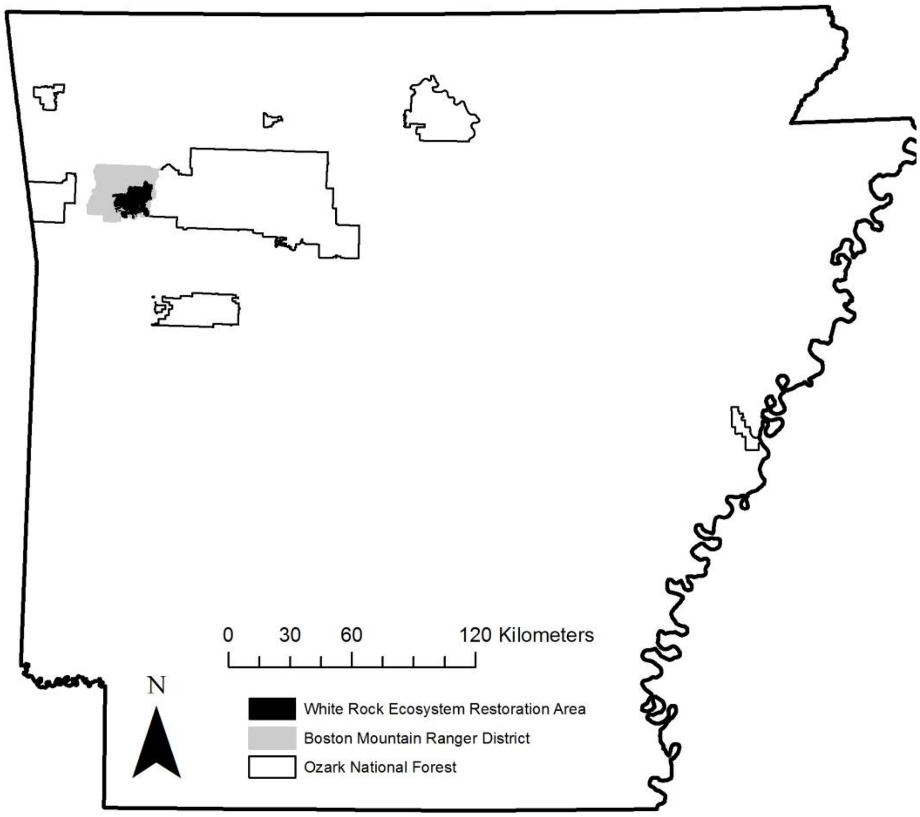

Fig 1. White Rock Ecosystem Restoration Area, Boston Mountain Ranger District, and the Ozark National Forest.

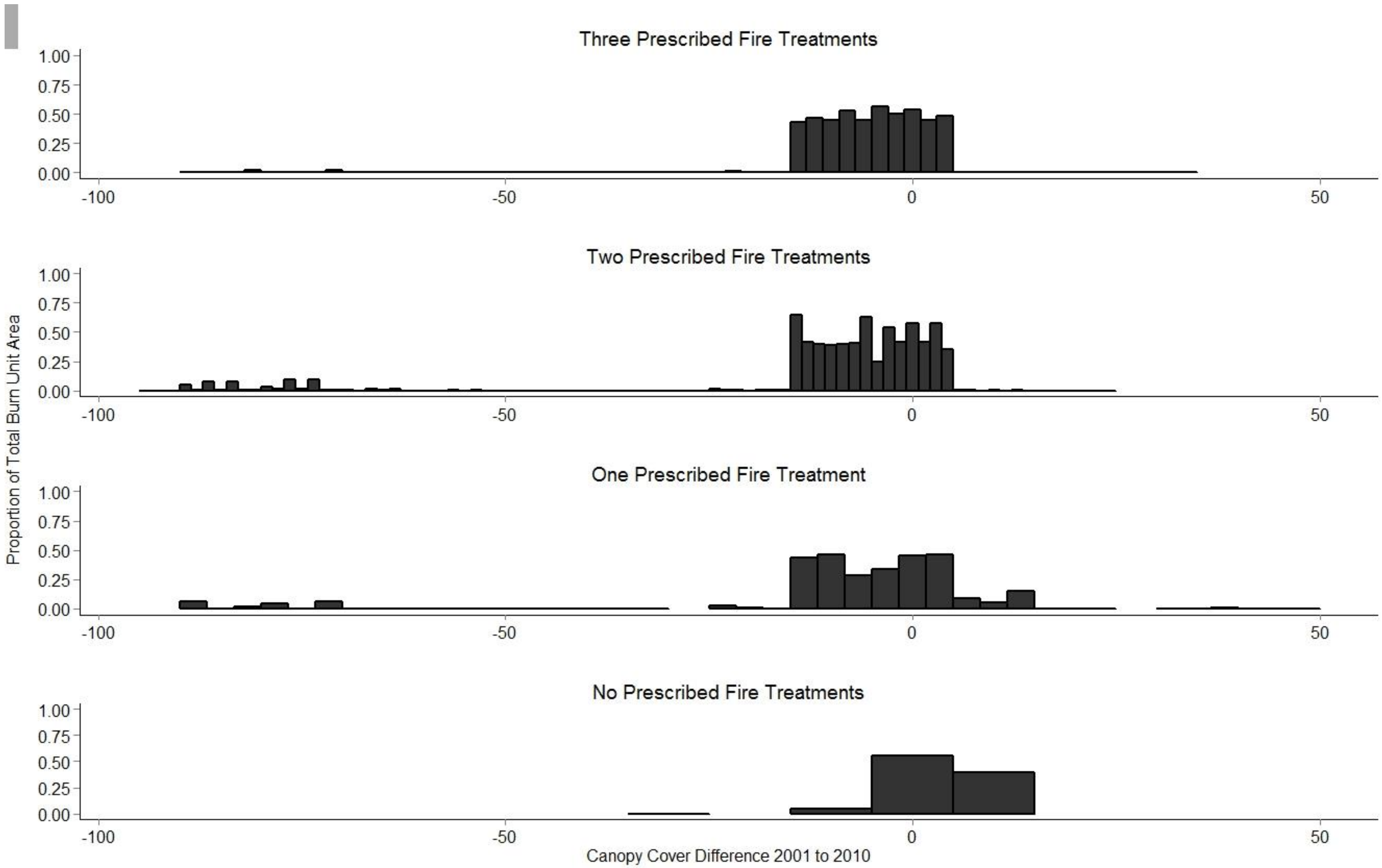

Fig 2. Change in percent canopy cover (from 2001 to 2010) on 15 prescribed fire units on the White Rock Ecosystem Restoration Area, Arkansas, USA. Units are divided based on the number of prescribed fire treatments received (0 treatments, n=1; 1 treatment, n=3; 2 treatments, n=7; 3 treatments, n=5).

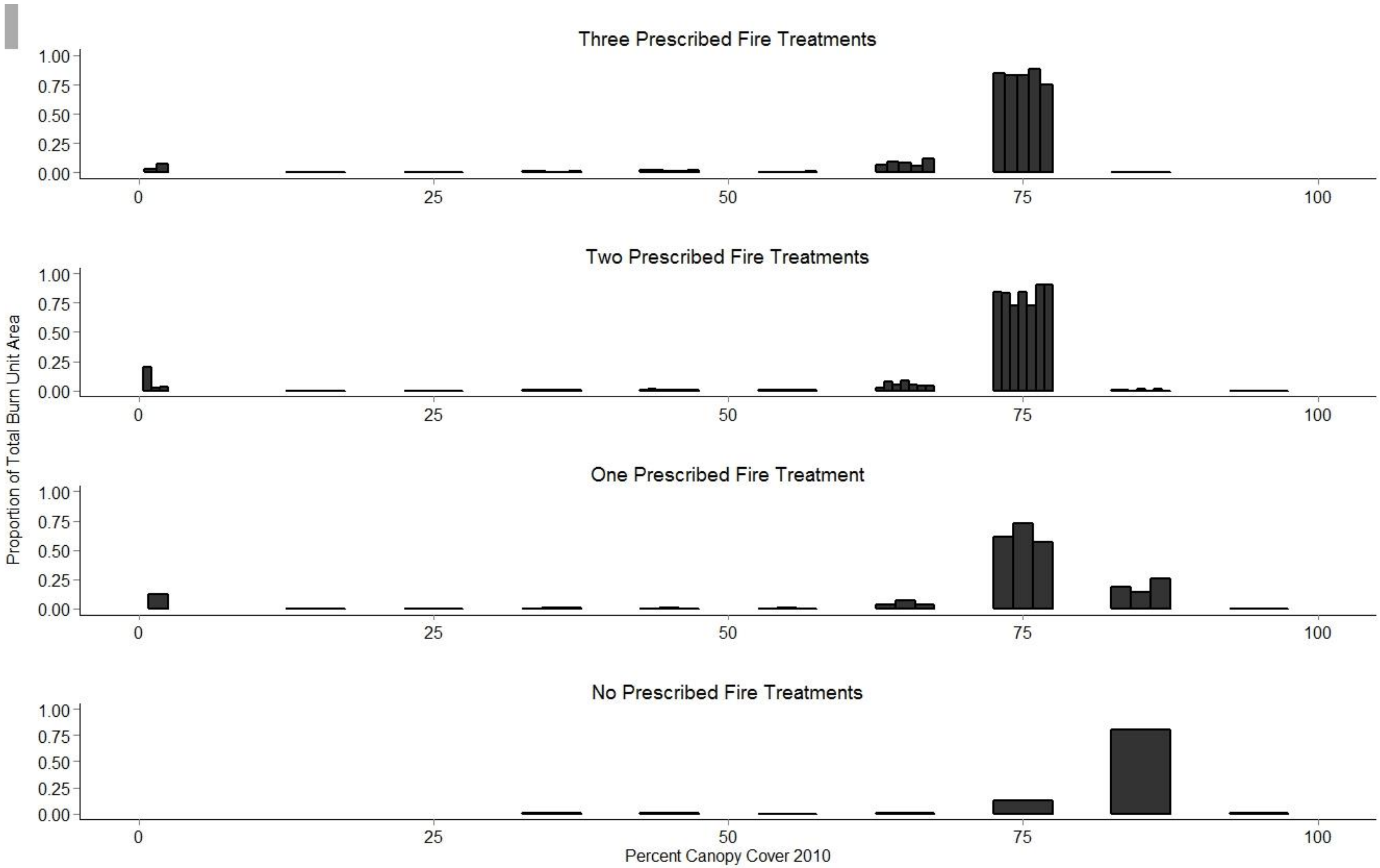

Fig 3. Percent canopy cover in 2010 on 15 prescribed fire units on the White Rock Ecosystem Restoration Area, Arkansas, USA. Units are divided based on the number of prescribed fire treatments received (0 treatments, n=1; 1 treatment, n=3; 2 treatments, n=7; 3 treatments, n=5).

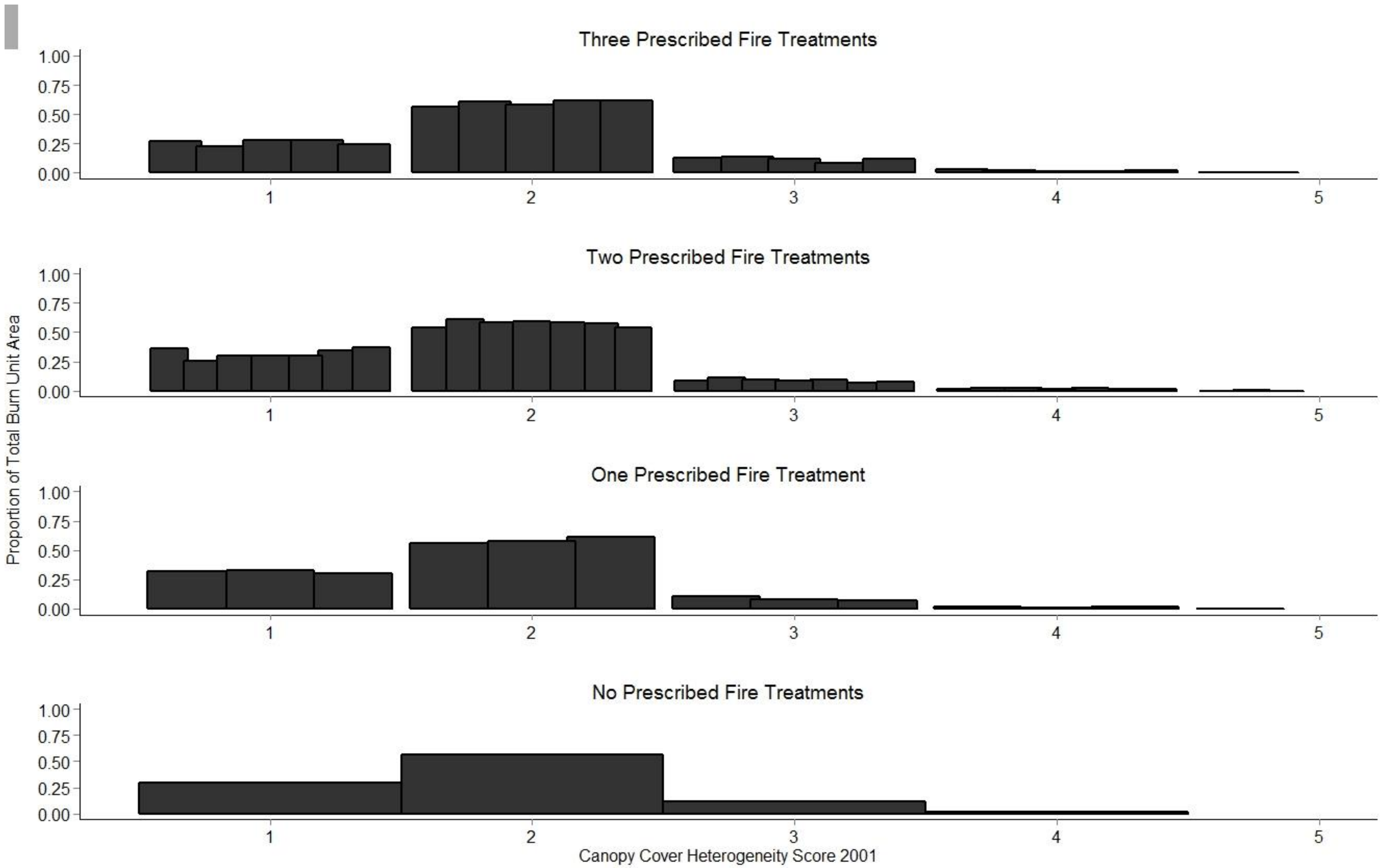

Fig 4. Canopy cover heterogeneity score in 2001 on 15 prescribed fire units on the White Rock Ecosystem Restoration Area, Arkansas, USA. U Although units are divided based on the number of prescribed fire treatments received (0 treatments, n=1; 1 treatment, n=3; 2 treatments, n=7; 3 treatments, n=5) no treatments had occurred in 2001.

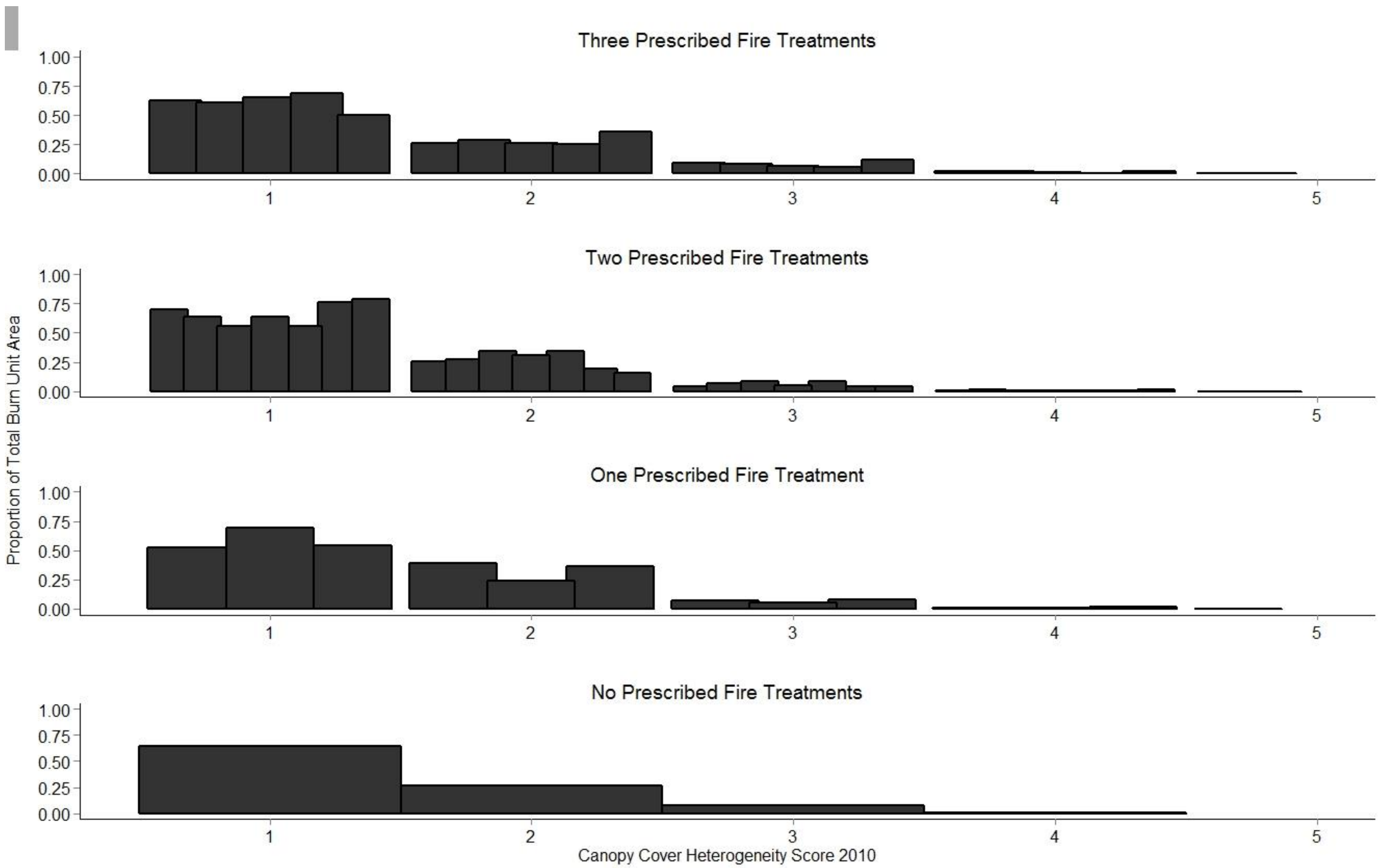

Fig 5. Canopy cover heterogeneity score in 2010 prescribed fire units on the White Rock Ecosystem Restoration Area, Arkansas, USA. Units are divided based on the number of prescribed fire treatments received (0 treatments, n=1; 1 treatment, n=3; 2 treatments, n=7; 3 treatments, n=5).

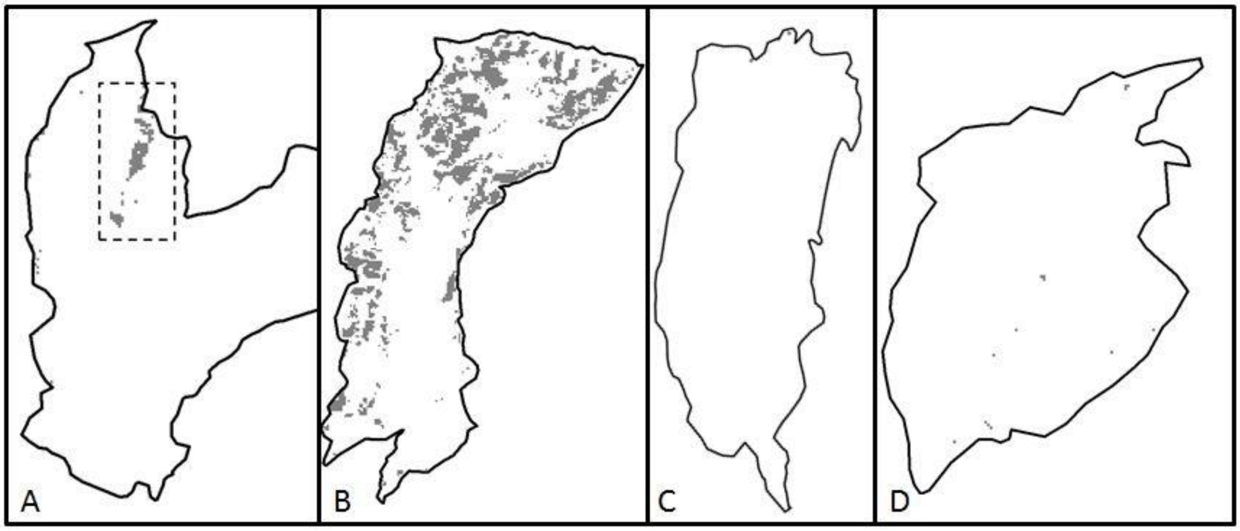

Fig 6. Change in canopy cover (from 2001 to 2010) for four prescribed burn units on the White Rock Ecosystem Restoration Area, Arkansas, USA under varying condition and treatments. Grey areas represent ≥ 25% reduction in canopy cover and white indicates a ≤ 25% reduction in canopy cover. Unit A (1403 ha) shows the signature of an area treated by a mechanical canopy reduction (dashed outline). Unit B (1665 ha) shows the signature of a natural canopy reduction (i.e. Insect damage, ice damage, etc.). Unit C (483 ha) shows no mechanical or prescribed burn treatments. Unit D (1012 ha) shows only 3 prescribed burn treatments. Individual pixels represent a 30 x 30 m area.

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Chapter 3, “Efficacy of landscape scale woodland and savanna restoration at multiple spatial and temporal scales” of H. T. Pittman’s dissertation is intended for submission for publication with one coauthor, D. G. Krementz.

I, Dr. David G. Krementz, advisor of Henry Tyler Pittman, confirm Henry Tyler Pittman will be first author and completed at least 51% of the work for this manuscript.

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Conclusion

Landscape level woodland and savanna restoration is becoming an increasingly common management strategy in the Central Hardwoods region and has been implemented on the White Rock Ecosystem Restoration Area (WRERA) of the Ozark-St. Francis National Forest in Arkansas since 2002. Management has primarily consisted of landscape scale (500-2500 ha) early growing season prescribed fire and has correlated with declines in eastern wild turkey (*Meleagris gallopavo* silvestris). We initiated this study since it is believed restoration has impacted the quality and/or quantity of wild turkey nesting habitat on the WRERA. Habitat quality and quantity has been suggested as factors limiting wild turkey populations (Thogmartin 1998). We captured and fitted 67 female wild turkey with 110 g Global Positioning System (GPS) Platform Transmitting Terminals between 2012 and 2013 on the WRERA to examine nesting and movement ecology in response to woodland and savanna restoration.

We used habitat data collected from 49 initial nest attempts and 16 renest attempts from 2012 to 2013, to determine habitat characteristics that discriminated nest-sites from 521 available unused nest-sites. We found that hens selected initial nest-sites with higher visual concealment (0-1 m in height), percent slope, and woody ground cover. We also found as time since prescribed fire increased so did visual concealment in a curvilinear fashion at initial nest-sites. Renest-sites were best characterized by higher visual concealment (0-1 m) and fewer small shrubs. Nest sites were placed in patches smaller than 40 m based on visual concealment (0-1 m). Hens also selected habitat in a hierarchical fashion. They selected habitat with higher canopy cover diversity and habitat that was in a transitional state, increasing canopy cover and vegetation height, within 100 m of their nest-sites.

We used 65 wild turkey nests from 2012 to 2013 to estimate productivity on WRERA. We estimated a nest success rate of 0.19 (95% CI [0.11, 0.31]) similar to estimates of Badyaev (1994), but among the lowest reported across the sub-species range. We also found that nest survival increased as percent visual concealment (0-1 m) increased, and nest survival decreased as the distance from a road increased. It is likely that both of these variables influence predator cues or the accumulation of those cues which affect nest survival. Thus hens selected nest-sites to maximize or minimize these criteria.

We examined habitat selection during the pre-incubation period using 1,469 unique choice sets from hens during the pre-incubation period at multiple spatial scales from 2012 to 2013. We examined pre-incubation habitat selection because increased amounts of pre-incubation habitat sampling have been positively correlated with productivity (Badyaev 1994). We found at the point scale hens selected habitat with increasing canopy cover and vegetation height from 2001 to 2010, higher values of canopy cover, and high canopy cover heterogeneity surrounding the point. At the 200 m and 400 m scales, hens selected habitat with higher values of canopy cover indicating the selection of habitat with vertical cover. Over all scales we found canopy cover, change in canopy cover and vegetation height from 2001 to 2010, and canopy cover heterogeneity were variables involved in the habitat sampling process. Our findings indicate that hens are sampling structurally diverse habitat during their search for quality nesting habitat.

We used 51,015 GPS locations of hens to estimate annual and seasonal home ranges from 2012 to 2013, to better understand hen movement ecology and habitat quality. We estimated mean 95% home ranges for adults, 2,703 ha (SE = 283.4) in 2012 and 4,750.4 ha (SE = 946.2) in 2013. Sub-adult 95% home ranges were 1,999.6 ha (SE = 440.5) in 2012 and 4,169.5 ha (SE =

751.7) in 2013. Home range estimates on average were over 100% larger than mean home range estimates previously documented in Arkansas (Badyaev 1994; Thogmartin 1998). We also estimated mean 50% core use areas of adults, 387.7 ha (SE = 29.4) and 373.3 ha (SE = 38.2) in 2012 and 2013 respectively, and sub-adults, 355.2 ha (SE = 74.5) and 358.2 ha (SE = 92) in 2012 and 2013 respectively. Core use areas were similar in size to those previously documented.

We examined the hypothesis that increased pre-incubation ranges are a characteristic of successfully nesting hens. Our results indicate this relationship holds in 2012 where pre-incubation ranges on average were larger for successfully nesting hens than those that were unsuccessful. However, in 2013 this relationship was opposite suggesting that other factors such as winter severity or variations in annual habitat quality, quantity or its spatial distribution could be influencing both pre-incubation ranges and nest success. Our results suggest that a lack of quality habitat is the cause of the large home and seasonal ranges of hens on WRERA. Our results also indicate that this is more specifically due to a lack of quality nesting habitat, causing increased habitat sampling during the pre-incubation period especially in sub-adults, since pre-incubation ranges were on average 46% of the annual home range area for hens on WRERA. These results suggest that landscape level early growing season prescribed fire for woodland and savanna restoration has not increased the amount of quality nesting habitat.

We used vegetation data collected at 70 locations within and around WRERA to examine the response of vegetation to early growing season prescribed fire as currently used for woodland and savanna restoration. We found the number of large shrubs was negatively related and small shrubs were positively related to prescribed fire severity. We also found that visual concealment from ground level to one meter in height was positively related to time since prescribed fire and woody ground cover was negatively related to the number of prescribed fire treatments. We also observed an effect of drought on visual concealment in units not treated with prescribed fire and those burned that year. We found that units treated with prescribed fire had higher forb and grass ground cover but not to levels characteristic of woodlands and savannas. Overall, we found that early growing season prescribed fire did change vegetation characteristics in years following treatment but did not achieve woodland and savanna characteristics.

We used LANDFIRE datasets from 2001 and 2010 to assess and quantify the efficacy of woodland and savanna restoration on WRERA since its implementation in 2002. We documented that after 8 years of woodland and savanna restoration with early growing season prescribed fire less than 5% (~1,000 ha) of the total area had achieved canopy cover consistent with woodlands or savannas. Overall canopy cover had been reduced but not enough to achieve woodland and savanna condition. Even after four prescribed fire treatments in some units canopy cover had not been reduced to woodland or savanna conditions. We also found that overall canopy cover heterogeneity decreased in units treated with prescribed fire. Across the WRERA early growing season prescribed fire intended for woodland and savanna restoration has not increased landscape diversity nor achieved woodland and savanna conditions as currently implemented after 8 years. To achieve significant changes in forest structure through the use of only early growing season prescribed fire management durations will range from 25 years on the most ideal sites to greater than 100 years on sites less suitable for woodland and savanna restoration.

In conclusion hens selected nest-sites with high amounts of visual concealment. However since woodland and savanna restoration began they have had to increase their space use in search of quality nesting habitat and in turn likely decrease their reproductive fitness. Increased space use and nest-site searching are reflected in productivity levels well below those of other wild turkey populations. We also observed that hens are attempting fewer renests after the failure of their initial nest attempt since woodland and savanna restoration began. In general our findings suggest that early growing season prescribed fire for woodland and savanna restoration has not increased the quality or quantity of habitat required by hens nor has it had the desired restoration outcomes. We also can conclude base on the increased energy expenditure associated with increase home range sizes and decreased renesting rates compared to before woodland and savanna restoration that overall productivity has decreased in the wild turkey population.

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