12-2012

Effects of Prescribed Fire and Habitat on Golden-winged Warbler (Vermivora chrysoptera) Abundance and Nest Survival in the Cumberland Mountains of Tennessee

Katie Lee Percy
kpercy@utk.edu

Recommended Citation

Percy, Katie Lee, "Effects of Prescribed Fire and Habitat on Golden-winged Warbler (Vermivora chrysoptera) Abundance and Nest Survival in the Cumberland Mountains of Tennessee. " Master's Thesis, University of Tennessee, 2012.
https://trace.tennessee.edu/utk_gradthes/1395

To the Graduate Council:

I am submitting herewith a thesis written by Katie Lee Percy entitled "Effects of Prescribed Fire and Habitat on Golden-winged Warbler (Vermivora chrysoptera) Abundance and Nest Survival in the Cumberland Mountains of Tennessee." I have examined the final electronic copy of this thesis for form and content and recommend that it be accepted in partial fulfillment of the requirements for the degree of Master of Science, with a major in Wildlife and Fisheries Science.

David A. Buehler, Major Professor

We have read this thesis and recommend its acceptance:

Joseph D. Clark, Craig A. Harper

Accepted for the Council:
Carolyn R. Hodges

Vice Provost and Dean of the Graduate School

(Original signatures are on file with official student records.)

Effects of Prescribed Fire and Habitat on Golden-winged Warbler (*Vermivora chrysoptera*) Abundance and Nest Survival in the Cumberland Mountains of Tennessee

A Thesis Presented for

The Master of Science

Degree

The University of Tennessee, Knoxville

Katie Lee Percy

December 2012

ACKNOWLEDGEMENTS

Graduate school has been a rewarding, but perhaps more importantly, humbling experience. Because indeed, the more I learn, the more I learn how little I know. Still, I am comforted by this concept, as a career in wildlife biology is guaranteed to be perpetually stimulating. My journey to this current state of mind has been facilitated by many, to whom I give acknowledgement.

Foremost, I would like to thank my major advisor, Dr. David Buehler, for the opportunity to be one of his graduate research assistants. It was a privilege and an honor to study the population of Golden-winged Warblers located in the Cumberland Mountains of Tennessee. The experience was invaluable, and exceeded my expectations for master's research in wildlife biology. I must also commend Dr. Buehler for his patience when discernible progress on my thesis slowed to a snail's pace. To Dr. Lesley Bulluck and Kelly Caruso I am indebted, for they documented the population of Golden-winged Warblers in the Cumberland Mountains Wildlife Management Area before me, laying a great deal of ground work that subsequently led to my thesis research. To my graduate committee members, Drs. Craig Harper and Joseph Clark, I have endless gratitude for their guidance during my studies at the University of Tennessee and valuable critiques of my thesis. It was an honor to have both serve on my committee, as they are model wildlife biologists. Also within the realm of academia, I extend acknowledgments to my undergraduate professors in the School of Renewable Natural Resources at Louisiana State University. It was through them that I was first introduced – however fateful – to the professional field of wildlife biology.

I alone could not have collected the amount of data necessary for my thesis research. For their hard work, I would like to recognize each and every technician who assisted this project from 2009 to 2011: Joel Kutylowski, Simone Hickman, Erica Rutherford, Matt Timpf, Mary Clapp, Sarah Miller, Jessie Seale, Emily Tewes, Ashley Jensen, and Matt Boone. I owe additional thanks to Joel Kutylowski and Matt Timpf, for their repeated field seasons and stellar nest searching abilities. Together we left blood, sweet and even a few tears on the Cumberland Mountains of Tennessee. I would also like to thank the Cornell Lab of Ornithology for their assistance with mtDNA analysis.

This research would not have been possible without financial support. Funding sources included: University of Tennessee, National Fish and Wildlife Foundation, American Bird Conservancy, Cornell Lab of Ornithology, U.S. Fish and Wildlife Service, Tennessee Wildlife Resources Agency, Kentucky Department of Wildlife and Fisheries, The Garden Club of America, Tennessee Ornithological Society, and The Nature Conservancy.

With respect to family, I am grateful for my parents, Adrian and Deborah Percy, who instilled in me an appreciation for the great outdoors. And it is with Brennan Percy, my twin brother and best friend, that I most enjoy sharing (and competing) in outdoor recreation. My entire family provided unwavering support during my pursuit of a master’s degree; they may never know the extent to which their love and encouragement eased my emotions during times of stress that tend to embody the graduate school experience.

Finally to my fellow graduate students within the Department of Forestry, Wildlife and Fisheries at the University of Tennessee, to whom I most graciously thank for their supportive friendships throughout this processes. Graduate school would have been an entirely different and lonely experience had it not been for their camaraderie. It is with utmost sincerity that I hope to maintain these relationships throughout the course of my lifetime.

ABSTRACT

Golden-winged Warbler (*Vermivora chrysoptera*) populations in Appalachia have declined precipitously over the past 45 years. The primary objective of my study was to monitor the response of Golden-winged Warblers to prescribed fire treatments on reclaimed coal mines in the North Cumberland Wildlife Management Area, Tennessee. Presence-absence surveys were conducted on eight mountain-top study sites and nest searching/monitoring was conducted on two additional sites, 2009-2011. I expanded on previous research of Golden-winged Warbler territory and nest-site selection by determining differences within main effects between used and unused territory plots, as well as used and unused nest-sites. Of my two nest-searched sites, Ash Log and Massengale Mountains, only Massengale received annual prescribed fire treatments, 2007-2011. Thus, analysis was conducted separately for these two study sites. I also modeled the effects of fire history, as well as temporal and biotic factors, on the variation in daily nest survival rates (DSR). I documented a population increase on Massengale, and no change on four sites. Population decline on three unmanaged sites was correlated with a decrease in shrub and/or *Rubus* spp. cover, and an increase in sapling height. Territories contained more shrub cover >1 m in height on Massengale and *Rubus* spp. cover was greater inside territories than on unused plots on Ash Log. No nest-plot variables differed between nest and non-nest plots. The best-supported model of DSR included the effect of year, quadratic time, and the presence of *Rubus* spp. in nesting substrate. Nesting success was highly variable across years; 10.8 ± 5.4% in 2009, 57.5 ± 8.8% in 2010, and 29.3 ± 10.0% in 2011. With respect to time, nest survival was greatest during peak of nest initiation in early May, declined through the middle of the nesting season, and increased again through the latter half of the season (27 June). Nest success decreased with the presence of *Rubus* spp. in the nesting substrate. I detected no negative relationship between daily nest survival and fire history. My study suggests that prescribed

burning on reclaimed coal mining land is a viable management practice for the creation and maintenance of Golden-winged Warbler breeding habitat.

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INTRODUCTION

The Golden-winged Warbler (*Vermivora chrysoptera*; hereafter Golden-winged) is a Nearctic-Neotropical migratory species experiencing precipitous population declines that are cause for significant conservation concern (Buehler et al. 2007). Thus, in February 2010, the Golden-winged was petitioned to be listed as threatened under the U.S. Endangered Species Act, and the status review was in progress upon the completion of this research (Sewell 2010). Population declines correlate with loss of migratory and wintering habitat, loss of suitable nesting habitat across their breeding range, competition and hybridization with Blue-winged Warblers (*Vermivora cyanoptera*, hereafter Blue-winged), and nest parasitism by Brown-headed Cowbirds (*Molothrus ater*) (Confer 1992, Buehler et al. 2007). In Tennessee, where population decline is particularly pronounced, nest parasitism has been rare and Blue-winged populations remain largely allopatric from Golden-wingeds (Welton 2003, Bulluck and Buehler 2008). Hence, the focus within this area has been on breeding habitat.

Golden-wingeds breed in early successional habitat, characterized by an herbaceous ground layer with some shrub and sapling cover, as well as a forested edge (Confer 1992, Klaus and Buehler 2001, Bulluck and Buehler 2008, Patton 2010). Historically, early succession plant communities resulted from windstorms, beavers, large grazing herbivores, and fire (Hunter et al. 2001, Lorimer 2001). In eastern North America, loss of early succession habitat is attributed in part to fire suppression, as well as farm land abandonment and forest maturation following extensive land clearing of the late nineteenth and early twentieth centuries. Currently within the Appalachian region, contour and mountain-top removal mining have created linear openings and larger expanses of grasslands and/or shrublands in an otherwise forested landscape (Canterbury et al. 1996, Patton et al. 2004, Bulluck and Buehler 2008).

Although Golden-wingeds are known to occupy reclaimed strip mines throughout the Cumberland Mountains of eastern Tennessee, it is unknown how long reclaimed mine land will provide suitable nesting habitat for Golden-wingeds without additional disturbance to maintain vegetation in early seral stages. It is also unknown to what extent management can restore and maintain suitable nesting cover on reclaimed coal mine lands in this region. The Tennessee Wildlife Resources Agency (TWRA) has the stated goal of developing and maintaining early succession habitat in the North Cumberland Wildlife Management Area (NCWMA). Specifically, TWRA has implemented prescribed fire treatments on reclaimed coal mine sites where Golden-wingeds are known to breed. However, few studies have examined the effect of fire on Golden-winged breeding habitat (Klaus 2004, Klaus et al. 2010). Thus, the primary objective of my study was to monitor the response of Golden-winged Warblers to prescribed fire treatments on reclaimed coal mines in the North Cumberland Wildlife Management Area, Tennessee. In Chapter 1, I assessed changes in Golden-winged, Blue-winged, and Golden-winged/Blue-winged hybrid abundances on 10 mountain-top study sites of both managed and unmanaged land. In Chapter 2, I evaluated changes in vegetation over time on unmanaged sites and modeled the effects of prescribed fire, as well as temporal and biotic factors, on daily nesting survival rates.

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CHAPTER 1: GOLDEN-WINGED WARBLER POPULATION MONITORING IN THE CUMBERLAND MOUNTAINS OF TENNESSEE

Abstract

I monitored the Golden-winged Warbler population on 10 mountain-top study sites in the North Cumberland Wildlife Management Area, Tennessee, 2009-2011, for changes in abundance, hybridization with Blue-winged Warblers (*Vermivora cyanoptera*), and nesting success. Probability of detecting Golden-winged Warblers during presence-absence surveys using playback recordings was 67.4 ± 5.1% and 50.6 ± 10.4% in early- and mid-June, respectively, when most surveys were conducted. After applying an average detection probability correction factor of 0.59 to abundance counts from presence-absence surveys, I documented a population decline on five sites, no change on four sites, and an increase on a single site which had been managed with prescribed fire. Overall, the population remained relatively stable with 96–108 breeding pairs on my ten study sites in 2011. Furthermore, I detected no increase in either Blue-winged Warblers or Blue-winged/Golden-winged hybrids. Across all 10 study sites, the average number of Blue-winged Warblers and phenotypic hybrids detected 2009-2011 was 4.3 and 10.3, respectively. During 2010 and 2011, mitochondrial DNA (mtDNA) analysis was conducted on individuals that phenotypically appeared to be pure Golden-wingeds. Of the 39 males and 12 females tested, four (10.3%) and zero, respectively, possessed the ancestral Blue-winged mtDNA haplotype. I used the logistic-exposure method in Program MARK to obtain estimates of daily nest survival rates. Nesting success was highly variable across years; 10.8 ± 5.4% in 2009, 57.5 ± 8.8% in 2010, and 29.3 ± 10.0% in 2011. Large annual fluctuations in reproductive success on my study sites highlight the need to establish long-term population trends and monitor source-sink dynamics.

Introduction

The Golden-winged Warbler (hereafter Golden-winged) is a Nearctic-Neotropical migratory species experiencing precipitous population declines over the last 45 years. Non-breeding range includes Central America and northern South America (Confer 1992). Golden-wingeds breed in eastern North America, from as far south as northern Georgia, northeast to Massachusetts and Quebec, and westward through the Great Lakes region and into Ontario, Manitoba and Saskatchewan (Buehler et al. 2007).

Based on analysis of North American Breeding Bird Survey data, Golden-winged populations have declined from 1966 to 2008 by 2.4% per year (C.I. = 3.4 – 1.5, n = 283 routes; Sauer et al. 2011), resulting in a 67% reduction since 1966. Golden-winged population declines correlate with loss of suitable nesting habitat across their breeding range, loss of wintering habitat, competition and hybridization with Blue-winged Warblers (*Vermivora cyanoptera*), and nest parasitism by Brown-headed Cowbirds (*Molothrus ater*) (Confer 1992).

As a result of population declines, the Golden-winged was petitioned to be listed as threatened under the U.S. Endangered Species Act in February of 2010 (Sewell 2010). The 90-day finding by the U.S. Fish and Wildlife Service (USFWS) concluded that the petition presented substantial scientific or commercial information indicating that the listing may be warranted (Federal Register 2011). Furthermore, the finding stated that a status review of the species will be initiated, and a 12-month finding will address whether listing is warranted.

In the meantime, Golden-wingeds have been declared a “species of national conservation concern” by the USFWS (2002), and “threatened” by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC 2006). The species is also on the Partners-in-Flight Continental Watch List (Rich et al. 2004), as well as the Audubon Society’s Watch List (Butcher et al. 2007). Individual state listings include “Endangered” in Indiana, Ohio, and Massachusetts, "Threatened" in Kentucky and Georgia, and "Of Management Concern" in Tennessee, New Jersey, New York, Connecticut, and Wisconsin (Buehler et al. 2007). Most recently, the Working Lands for Wildlife, a new partnership between U.S. Department of Agriculture's Natural Resource Conservation Service and the USFWS, will use "agency technical expertise combined with $33 million in financial assistance from the Wildlife Habitat Incentive Program (WHIP)" to promote Golden-winged conservation practices, along with six other focal species (NRCS 2012).

The Golden-winged population decline is particularly significant in the Appalachian Mountains Bird Conservation Region (-8.3% per year; C.I. = 9.6 – 7.0; n = 108 routes). In Tennessee the estimated rate of decline is -7.7% per year (C.I. = 11.5 – 4.4), but Golden-wingeds have been encountered on only 7 out of 42 routes throughout the state during the monitoring period, thus the reliability of the estimates is questionable (Sauer et al. 2011).

Golden-wingeds occur in the Cumberland Mountains of Tennessee near the southernmost portion of their breeding range. Such rear-edge populations may be critically important to long-term conservation of genetic diversity, but under-studied in ecological research (Hampe and Petit 2005). Monitoring the population in this region began in 2000, with the Golden-winged Warbler Atlas project coordinated by the Cornell Laboratory of Ornithology (1999, Welton 2003). Research through the University of Tennessee was initiated in 2003, and the population has been monitored every year since, making this one of the longer term studies of the species in a specific area. Bulluck and Buehler (2008) estimated that 370 pairs of Golden-wingeds breed in the Cumberland Mountains of Tennessee, making this the largest population in the southern portion of its range.

Range expansion of the Blue-winged Warbler and subsequent hybridization with Golden-wingeds is thought to be a contributing cause of the Golden-winged population decline. Historical populations of Golden-wingeds and Blue-wingeds were apparently allopatric, with genetic divergence occurring ~1.5 million years ago (Gill 2004). Genetic data indicate the mitochondrial DNA (mtDNA) of Golden-wingeds and Blue-wingeds differ by about 3 – 4.5% (Gill 1997, Shapiro et al. 2004, Dabrowski et al. 2005).

Land clearing following European settlement has facilitated Blue-winged expansion into the range of Golden-wingeds (Gill 1980, Hunter et al. 2001). In some areas (e.g. study sites in Pennsylvania), this may have led to the extirpation of Golden-wingeds, as a result of asymmetric introgression of Blue-winged DNA into Golden-winged populations following hybridization events (Gill 1997, Confer et al. 2003). However, more recent and extensive mtDNA surveys found bi-directional introgression between Golden-wingeds and Blue-wingeds at sites where both species were well represented (Shapiro et al. 2004, Dabrowski et al. 2005).

In the Cumberland Mountains of Tennessee, Golden-wingeds and Blue-wingeds are generally allopatric based on elevation (Welton 2003, Bulluck 2007). Welton (2003) documented greater numbers of Blue-wingeds at sites 331 to 458 m in elevation; Golden-wingeds became more abundant as elevation increased. No Blue-wingeds were documented above 843 m, though hybrids were detected at the upper elevational extreme (971-1098 m). As a result, Golden-winged-Blue-winged pairings are infrequent in this region. However, Vallender et al. (2009) reported a 14% mismatch of mitochondrial haplotype from the blood samples of 92 phenotypically pure Golden-wingeds collected in the Cumberland Mountains of Tennessee; thus, introgression is occurring and the Golden-winged population of the Cumberland Mountains is not entirely genetically pure. Hybridization is of particular importance when it involves threatened or endangered species because it degrades genetic integrity, and, moreover, exempts individuals from legal protection under some conservation regulations (Allendorf et al. 2001, Roth et al. 2012).

Study of the Golden-winged population in the Cumberland Mountains of eastern Tennessee began in 2003. Ultimately, a population viability analysis will be both necessary and possible from this long-term dataset. Population abundance, reproductive success, and survival are fundamental to the study of population dynamics in ecology. This information is critically important to conservation projects attempting to recover and/or maintain the viability of threatened populations. With that in mind, my objectives were to conduct a general population status assessment by 1) tracking changes in abundance of Golden-wingeds across 10 study sites, 2) monitoring the phenotypic and genotypic rate of hybridization and 3) searching for and monitoring nests to determine daily nest survival and reproductive success. I hypothesized that a population decline is occurring on my study sites that have not received management intervention to arrest succession of vegetation. Assuming an elevational separation remains between the Golden-winged and Blue-winged populations of Tennessee, I hypothesized that the percent of Golden-winged-Blue-winged hybrids on my study sites has not change, relative to 2003-2005 estimates. Because fluctuations in predator communities and/or weather patterns may occur annually, I hypothesized that annual variation in Golden-winged nest survival rates would occur.

Methods

Study area

The study sites covered ten mountain-tops of both managed (e.g., prescribed fire, timber harvest, vegetation thinning, grass and forb seeding) and unmanaged land located within and adjacent to the North Cumberland Wildlife Management Area (Fig. 1.1). Sites were selected in

2003 and 2005 because they had known or expected concentrations of breeding Golden-wingeds (Welton 2003) including 1) Ash Log Mtn., 2) Massengale Mtn., 3) Red Oak Mtn., 4) Bootjack Mtn., 5) Fork Mtn., 6) Burge Mtn., 7) McNew Gap, 8) Brushy Mtn., 9) Hatfield Knob and 10) Anderson Mtn. I concentrated nest searching on only Ash Log and Massengale Mountains although other sites had nest monitoring in previous years.

The mean elevation of all ten sites was 800 m (range = 650-960 m). Coal surface-mine reclamation occurred from approximately 1980 to 1990 (Bulluck and Buehler 2008). Reclamation involved planting black locust (*Robinia pseudoacacia*), tall fescue (*Festuca* arundinacea), sericea lespedeza *(Lespedeza cuneata)*, and autumn olive (*Elaeagnus umbellata*). Yellow-poplar (*Liriodendron tulipifera*), maples (*Acer* spp.), oaks (*Quercus* spp.), blackberry (*Rubus* spp.), and a variety of forbs (e.g. *Solidago* spp.) and grasses (e.g. orchard grass, *Dactylis glomerata*) have since colonized the sites.

Presence-absence surveys

I conducted surveys for Golden-wingeds, Blue-wingeds, and their hybrids on the eight study sites where nest searching did not occur. Surveys were conducted between early May and June 15th – the main breeding season of Golden-wingeds in Tennessee (Bulluck and Buehler 2008). Each survey was conducted between sunrise and 1100, as detectability of passerines generally decreases later in the day (IBCC 1970). Surveys were not conducted during inclement weather, such as precipitation or moderate-to-heavy wind that would decrease detection rates (IBCC 1970).

I walked transects along mining and logging roads along strip benches at a slow pace while listening and watching for the species. Playback recordings of Golden-winged and Blue-winged song (Type I and Type II; Ficken and Ficken 1967) were broadcast to elicit a response from territorial birds and enhance the probability of detection (Ficken and Ficken 1973, Johnson et al. 1981, Cornell Lab of Ornithology 1999, Kebel and Yahner 2007). Kubel and Yahner (2007) found detectability of Golden-wingeds decreased beyond a distance of 100 to 150 m, especially in shrub-land forest areas, similar to our study sites. In an effort to enhance detection, I broadcasted recorded territorial songs on a MP3 player with a portable speaker every 100 to 150 m.

Each stop along the survey route began with a 2-min pre-playback observation period (i.e., no broadcasting). The pre-playback period was followed by a 3-min period of playback, consisting of 1-min Golden-winged Type 1 song, 1-min of Golden-winged Type II song, and lastly 1-min of Blue-winged Type 1 song. A 2-min post-playback observation period followed before walking 100-150 m to the next playback stop. Because Golden-wingeds and Blue-wingeds can sing each other's song, auditory detection alone does not definitively identify the species (Ficken and Ficken 1967, Confer 1992). Thus, visually sighting individuals is required. Combining pre-playback, playback, and post-playback periods in this manner further increases Golden-winged detection; mean (±SE) detection probabilities using this protocol were 81.9 ± 4.1% and 33.4 ± 4.9% in right-of-ways and young (1-3 yr) clearcuts, respectively, in Centre County, Pennsylvania (Kebel and Yahner 2007).

Detectability is an important metric to consider, especially when abundance estimates from presence-absence surveys are compared to abundance counts (e.g. from extensive nest searching). Therefore, a correction factor is needed to more accurately estimate the population. During the 2011 breeding season, I estimated detection probabilities (i.e., the percentage of known population detected) for male Golden-wingeds on our study sites by conducting presence-absence surveys (following the protocol outlined above) throughout the two mountain sites where extensive nest searching occurred (MacKenzie et al. 2005). Territories were delineated throughout the breeding season, and were based on a minimum of five observations. This was similar to the criteria used by Kubel and Yahner (2007) to designate known Golden-winged territories, as well as guidelines by the International Bird Census Committee (1970). Presence-absence surveys throughout territories were conducted twice at each sampling location. The first round of surveys occurred 05 – 08 June 2011; the second round of surveys occurred 10 – 15 June 2011. I calculated the probability of detection for each survey by summing all observations during the survey and dividing by the number of known territories. I estimated overall detection probability for a given time period as the mean of probabilities for all surveys combined (n = 4 for round 1 and round 2). Small sample sizes and a lack of replication within sampling period prevented more thorough statistical analysis.

Hybridization Estimation

I target mist-netted adult male and female Golden-wingeds for the collection of blood and feathers (P1, R3 or R1, and facial mask) during 2010 and 2011. MtDNA analysis was conducted by the Cornell Lab of Ornithology, Ithaca, NY. Individuals were assigned their ancestral haplotype following procedures outlined by Vallender et al. (2009). The University of Tennessee Animal Care and Use Committee approved the field methods used in this study (Protocol #561-1101).

Nest searching and monitoring

I searched for and monitored nests on two sites (Ash Log Mountain and Massengale Mountain) to determine daily nest survival and reproductive success. These sites were selected because Golden-winged abundance was high, vegetation was similar (coal mine reclamation occurred at approximately the same time) and prescribed fire was being used to restore and maintain early successional vegetation.

Each field technician was assigned specific areas (~40 ha in size) for the nesting season. This allowed them to become familiar with all Golden-winged, Blue-winged, and hybrid territories within their area, maximizing the number of nests located. The peak of the Golden-winged breeding season in the Cumberland Mountains lasts approximately six weeks (Bulluck and Buehler 2008). Nest searching began the last week in April and concluded by mid-June. Nest searching and monitoring followed the suggested guidelines of Martin and Geupel (1993). Precautions were taken to minimize the effect of human disturbance on nest abandonment or depredation. Once a nest was found, it was monitored every 1 to 4 days (for number of eggs and young) until fledging or confirmed nest failure occurred. Monitoring occurred daily as nests approached anticipated fledging days (~nestling day 7), to best determine nest fate and to count the number of nestlings present. Any nest that fledged ≥ 1 young was considered successful.

Analysis

Nest Survival

I used the nest survival model in Program MARK to obtain estimates of daily nest survival rates (DSR; White and Burnham 1999, Rotella et al. 2004). Successful and unsuccessful nests are not found with equal probability, which can easily bias apparent nesting success high, relative to actual nesting success (Mayfield 1961, 1975). Like Mayfield's method, the nest survival module in Program MARK accounts for finding nests at different stages of the nesting cycle.

Program MARK uses the logistic-exposure method to estimate the probability that a nest survives a single day. Using the logit link function, constraining survival to the binomial interval (0, 1), DSR of a nest on day i is modeled as

$$DSR = \frac{exp(\beta_o + \Sigma_j \beta_j x_{ji})}{1 + exp(\beta_o + \Sigma_j \beta_j x_{ji})}$$

where β_j are coefficients to be estimated and x_{ji} are values for *j* covariates on day *i* (Rotella 2010). DSR can then vary among groups of nests, among individual nests, and among days (Rotella et al. 2004). Parameters (*βj*) of each model are estimated iteratively (Rotella 2010). True probability of a nest surviving from initiation to completion (i.e., nest survival) is then calculated as $(DSR)^n$, where *n*, the total length of the nesting period, equals 25 days for Golden-wingeds (5 days laying + 11 days incubating + 9 days brooding) on our sites.

Data input requires five values for each nest: 1) the day the nest was found, 2) the last day the nest was checked when alive, 3) the last day the nest was checked, 4) the fate of the nest (0 = successful, 1 = depredated or abandoned), and 5) the number of nests that had this history. I used evidence of nest disturbance, presence of fecal sacs or eggshell fragments, and age of the nest to determine nest fate. I attempted to locate young if fledging was suspected. Golden-winged fledglings can be quick to disperse, however, leaving the male's territory within hours of the fledging event. In those cases, I monitored the behavior of the parents (e.g. carrying food) for indications of nest fate.

I obtained separate DSR estimates for nests of 1) phenotypically pure Golden-wingeds, 2) phenotypically pure male and/or female Blue-wingeds and 3) phenotypic male and/or female hybrids. To address issues of sample size, I pooled the nests of male and/or female hybrids in the dataset for modeling annual variation in DSR. I did, however, remove the nests of Blue-wingeds from the modeling dataset because differences in breeding habitat between the species have been documented on other study sites (Confer and Knapp 1981, Confer et al. 2003, Patton et al. 2010).

Results

Presence-absence surveys

Of 44 known territories on Ash Log and Massengale, 29 individual males were detected during the first round (05 – 08 June) of known presence-absence surveys; 20 individual males were detected during the second round (10 – 15 June). Mean (±SE) detection probabilities for the first and second round of surveys were 67.4 ± 5.1% and 50.6 ± 10.4%, respectively (Table 1.1). Using the range of detection probability values to adjust abundance counts, there were 96–108 breeding pairs on my ten study sites in 2011.

Abundance of Golden-wingeds varied across the ten sites from a low of zero breeding pairs on Red Oak, Brushy and Hatfield to a high of 38 breeding pairs on Ash Log in 2011 (Table 1.2). On Massengale Mountain, the Golden-winged population increased from approximately 8 breeding pairs in 2003 to 24 breeding pairs in 2011 (Table 1.2 and Figure 1.2). The population has remained relatively stable on Ash Log and Bootjack, largely in the absence of active habitat management (one 35-ha portion of Ash Log was burned in 2007; Bootjack was not burned). Conversely, population decline or extirpation has occurred on five survey sites (Fork Mtn., Burge Mtn., Red Oak Mtn., McNew Gap and Hatfield Knob; Table 1.2 and Figure 1.2).

Hybridization

Across 10 study sites, the proportion of Blue-wingeds detected during presence-absence surveys and nest searching was 5.8% (n = 5) in 2009, 3.8% (n = 3) in 2010, and 5.6% (n = 5) in 2011; the proportion of hybrids detected was 12.8% (n = 11) in 2009, 11.5% (n = 9) in 2010, and 12.2% (n = 11) in 2011 (Figure 1.3). During 2010 and 2011, mtDNA analysis was conducted for 39 males and 12 females that appeared, phenotypically, to be pure Golden-wingeds. Of the males tested, 10.3% possessed the ancestral Blue-winged mtDNA haplotype. No females possessed the ancestral Blue-winged mtDNA (Table 1.3).

Nest survival

I monitored 93 nests with ≥ 1 egg over three breeding seasons (2009-2011). Specifically, 44.1% were located during the nest construction stage, 34.4% were located during the laying/incubation stage and 21.5% were located during the nestling stage. Nests located during construction but never had an egg laid in them were excluded from nest survival analysis.

During 2009, 29 nests were monitored, consisting of 24 nest-pairs of phenotypically pure Golden-wingeds, 4 nest-pairs of a male and/or female hybrid, and 1 nest-pair included a male Blue-winged. During 2010, 37 nests were monitored, consisting of 26 nest-pairs of phenotypically pure Golden-wingeds, 10 nest-pairs of a male and/or female hybrid, and 1 hybrid male (who sang a Blue-winged song) nested with a female Blue-winged. During 2011, 27 nests were monitored, consisting of 20 nest-pairs of phenotypically pure Golden-wingeds, and 6 nest-pairs of a male and/or female hybrid. One of the hybrid males nested with a female Blue-winged. In 2011, one nest-pair consisted of male and female Blue-wingeds (Table 1.4).

The earliest documented nest initiation date (i.e. first date an egg was laid) was 2 May 2011 which was standardized as day 1 for DSR analysis, and all nest check dates were sequentially numbered thereafter. The last active nest date was 27 June 2011, resulting in a 57-day nesting season with a total of 1,072 exposure days. Approximately 95% of nests could be precisely aged (n = 88), based on either egg laying or aging of nestlings. Five nests were located during incubation that did not survive to hatching; hence their precise age could not be determined. I assigned these nests the median age of the average Golden-winged incubation stage.

Nests of phenotypically pure male and female Golden-wingeds (n = 70) had an estimated DSR ± SE of 0.9607 ± 0.007 (36.7 ± 6.4% success) and an average of 4.3 young fledged per successful nest; nests of a male and/or female Blue-winged (n = 4) had an estimated DSR ± SE of 0.939 ± 0.042 (20.9 ± 23.1% success) and an average of 5.0 young fledged per successful nest; and nests of a male and/or female hybrid (n = 19) had an estimated DSR ± SE of 0.9532 ± 0.013 (30.2 ± 10.4% success) and an average of 4.0 young fledged per successful nest (Table 1.5). Nesting success was highly variable across years; 10.8 ± 5.4% in 2009, 57.5 ± 8.8% in 2010, and 29.3 ± 10.0% in 2011 (Table 1.6).

Average clutch size was 5 eggs on the first nesting attempt. During 2010, I recorded one nest with 6 eggs. Clutch size decreased by 1 egg with each subsequent nesting attempt following a nest failure. Apparent nest success from 2009 to 2011 was 49.5% (46/93), with an average of 4.3 young fledged per successful nest. Of the 46 nests designated "failures," 44 (96%) were attributed to predation. Only three nests with eggs were documented as abandoned. Each of these three nests was subsequently depredated. I was unable to determine specific predator species per predation event.

Discussion

Presence-absence surveys

An apparent increase in breeding pairs detected from 2003 to 2005 may be attributed to refining our search image for Golden-wingeds at the onset of this research. Massengale Mountain is the only site where a population increase was documented over the entire course of monitoring. Massengale is also the only site where prescribed fire was implemented from 2006 through 2011, suggesting fire has been effective at restoring and maintaining Golden-winged breeding habitat on a reclaimed coal mine.

Largely in absence of active habitat management, the population has remained relatively stable on Ash Log and Bootjack Mountains. It remains of question when vegetation on these sites will proceed into later seral stages no longer suitable for Golden-winged nesting. I hypothesize that such is the situation on Fork, Burge, McNew Gap, Hatfield Knob and Red Oak mountains, where population decline has occurred.

Although our presence-absence survey method allowed us to survey the entire areas of interest, it is unlikely that 100% of male Golden-wingeds occupying our survey sites were detected, despite the use of playback (McShea and Rappole 1997, MacKenzie et al. 2005, Kubel and Yahner 2007). I speculated that foraging activity (particularly into mature forest), territorial interactions with neighboring males, and a general lack of territorial aggression later in the breeding season may have decreased detectability of Golden-wingeds during presence-absence surveys.

Detection probabilities should be calculated when estimating Golden-wingeds because they are commonly undetected (Kubel and Yahner 2007). Several studies have investigated Golden-winged detection rates, with and without the use of playback (Kubel and Yahner 2007, Aldinger 2010), and in relation to vegetation types (Kubel and Yahner 2007). A single range-wide correction factor is difficult to establish because detection may differ with differences in duration and type of song playback used (Aldinger 2010), time of season (Kubel and Yahner 2007), and habitat composition (Kubel and Yahner 2007).

Aldinger (2010) documented detection probabilities of male Golden-wingeds during three periods on grazed areas in the Monongahela National Forest, West Virginia. During their 10-21 May sampling period, mean detection was 76.1% with Type I playback and 59.9% with Type II playback. During their 22 May – 2 June sampling period, mean detection was 64.3% with Type I playback and 72.1% with Type II playback. During their 3 – 15 June sampling period, mean detection was 57.5% with Type I playback and 59.7% with Type II playback, similar to our results during the same time period. Our detectability methods were regionally consistent.

Kubel and Yahner (2007) conducted surveys with and without playback in 60-m-wide powerline rights-of-way (ROW) and habitat managed with 1-ha clearcuts (1-3 years old) surrounded by forest, located in Centre County, Pennsylvania. Combining pre-playback, playback, and post-playback, as I did for presences-absence surveys, mean detection probability was 81.9 ± 4.1% in the ROW and 33.4 ± 4.9% in the clearcuts. Surveys were conducted 20 May to 17 June, 2002 and 2003. Probabilities were estimated for three individual time periods (20–31 May, 1–10 June, and 11–17 June), but they did not find a consistent pattern in detection probabilities over time. Observations of male Golden-wingeds were distributed evenly across time periods during 2002 and skewed toward the earlier period during 2003. Similar to ROWs, reclaimed coal contour mines have linear strip benches. But unlike ROWs, the strip benches on my study sites were not completely open, and contain shrub and tree cover that may decrease detectability. Furthermore, our reclaimed coal mining sites are not solely linear; many sites encompass scrub-forest wider than a 60-m ROW.

Although my presence-absence surveys were conducted within a time period that was believed to encompass the peak of Golden-winged breeding activity within our study area (early May to mid-June), singing frequency and territorial defense by Golden-winged males appeared to decrease after 1 June. Average nest completion (successes and failures) was 2 June 2009-2011. Known-presence-absence surveys, conducted on my two intensive nest-searched sites (Ash Log and Massengale Mountains), documented an average probability of detection that ranged from 50.6 – 67.4 and decreased over time. These results were consistent with more rigorous studies of Golden-winged detection probabilities conducted within different regions of their breeding range (Kubel and Yahner 2007, Aldinger 2010).

Hybridization

Hybrid individuals may have reduced pairing success (Canterbury et al. 1996, Confer and Tupper 2000, but see Vallender et al. 2007), but hybrid pairs do not appear to have reduced reproductive success compared with Golden-winged pairs (Vallender et al. 2007). Furthermore, hybrid progeny are fertile (Canterbury et al. 1996, Confer and Tupper 2000). Our results are consistent with the literature. The overall nest success rate (2009-2011) among phenotypically pure Golden-wingeds was similar to that among pairs that included a male and/or female hybrid (36.7% and 30.2%, respectively), as was average clutch size (4.46 and 4.24, respectively) and average young fledged per successful nest (4.30 and 4.14, respectively).

Hybrids have been documented throughout the Golden-winged's breeding range, including Manitoba, Canada – once thought to be the sole remaining region of the breeding range where a genetically pure population of Golden-wingeds existed (Vallender et al. 2009). Gill (1980) documented a consistent pattern of Golden-winged disappearance from an area within a 50-year period of Blue-winged introduction and subsequent hybridization between the two species. More pronounced is the apparent complete replacement that occurred within five years of initial contact at river valley sites in southern West Virginia (Canterbury et al. 1996).

In contrast to the predictions by Gill (1980), species coexisted for at least a century in the Sterling Forest State Park of southern New York (Confer et al. 1998, Confer and Tupper 2000). Although long-term coexistence may prevail in some areas, the southern Appalachians and Cumberland Mountains may warrant conservation attention, as the two species appear to be largely segregated by elevation. Blue-winged warbler and hybrid abundance did not show a marked increase in the NCWMA over the course of my study. Nor was an increase in cryptic hybridization detected through molecular analysis relative to 2003-2005 results. Of 39 phenotypic Golden-winged males tested during 2010-2011, ~10% exhibited evidence of cryptic hybridization, slightly less than the ~14% detected out of 93 individual males tested during 2003-2005. Minimally, the rate of hybridization over time should continue to be monitored in this region, as it may further illuminate this hybridizing complex and dictate management action.

Although the use of maternally inherited mtDNA markers has improved our understanding of hybridization between Golden-winged and Blue-winged Warblers, it is limited by its inability to provide information about paternally-inherited genes. The inclusion of data from nuclear DNA, which is biparentally inherited, would extend our understanding of this hybridizing complex and make it possible to identify additional cryptic hybrids (i.e., individuals with mtDNA and phenotype matched but still containing levels of introgression).

Nest Survival

Nest success rates for Golden-wingeds have been variable across the breeding range. Based on studies that used either Mayfield or maximum-likelihood estimation techniques, nest success ranges from a low of 20% within utility ROWs in Pennsylvania (Kubel and Yahner 2008) to a high of 73% within forest regeneration areas (<20 yr) in the Nantahala National Forest, North Carolina (Klaus and Buehler 2001). Although my nest success rate of ~10% during 2009 is disconcerting, the success rate of ~55% during 2010 is among the highest reported across the entire breeding range (Canterbury et al. 1996, Demmons 2000, Confer et al. 2003). Additionally, average nest mortality rate documented on my study sites is consistent with Martin’s (1995) summarization of the literature, which reported birds nesting on the ground in shrub/grasslands experienced greater predation rates (48.8%) than ground nests in less disturbed forested habitats (30.6%). Large annual fluctuations in reproductive success on my study sites highlight the need to establish long-term population trends and monitor source-sink dynamics. A comprehensive population viability analysis should be considered to assess long term sustainability of the Golden-winged Warbler in the Cumberland Mountains of Tennessee.

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CHAPTER 2: THE INFLUENCE OF MANAGEMENT ON GOLDEN-WINGED WARBLER BREEDING HABITAT AND NEST SUCCESS IN THE CUMBERLAND MOUNTAINS OF TENNESSEE

Abstract

Based on analysis of North American Breeding Bird Survey data, Golden-winged Warbler populations have declined from 1966 to 2008 by 2.4% per year (C.I. = 3.4 – 1.5, n = 283 routes), resulting in a 67% reduction since 1966. The population decline is particularly significant in the Appalachian Mountains Bird Conservation Region (-8.3% per year; C.I. = 9.6 – 7.0; n = 108 routes). I studied a population of Golden-winged Warblers in the southernmost portion of their breeding range, where distribution is limited to higher elevation (>600 m). Ten mountain-top study sites were located in the North Cumberland Wildlife Management Area of northeastern Tennessee, 2009-2011. The population declined on three sites since 2004, and increased on one site where prescribed fire was implemented annually, 2006-2011. I re-measured vegetation inside nest-plots that were first measured in 2005-2006 on the three sites where a population decline occurred. I nest searched on two sites where the Golden-winged population was largest and prescribed fire was used to create/maintain early successional vegetation composition. I collected habitat measurements within plots centered on nests and the accompanying male's territory, and tested for differences between non-nest sites and unused territory plots, respectively. My two nest-searched study sites were analyzed separately because there were differences in vegetation, as management was active on only one site, 2006-2011. I also modeled the effects of temporal and biotic factors on the variation in daily nest survival rate (DSR). The decline in the Golden-winged population across three study sites was correlated with a decrease in shrub and/or *Rubus* spp. cover, as well as an increase in sapling height and canopy cover. Territories contained more shrub cover >1 m in height on the managed study site. Similarly, *Rubus* spp. cover was greater inside territories than on unused plots on the other nest-searched site. No nest-plot variables differed between nest and non-nest plots. The best-supported model of DSR included the effect of year, quadratic time, and the presence of *Rubus* spp. in nesting substrate. Nest success was highly variable across years; 10.8 ± 5.4% in 2009, 57.5 ± 8.8% in 2010, and 29.3 ± 10.0% in 2011. With respect to time, nest survival followed a quadratic curvilinear pattern, where survival was greatest during peak of nest initiation in early May, declined through the rest of May and into early June, and increased from approximately the second week in June through the remainder of the nesting season (27 June). Contrary to territory selection, DSR decreased with the presence of *Rubus* spp. in the nesting substrate. I detected no negative relationship between DSR and fire history. Our study suggests that prescribed burning on reclaimed coal mining land is a viable management practice for the creation and maintenance of Golden-winged Warbler breeding habitat.

Introduction

Golden-winged Warbler nesting habitat is a patchy mosaic of early-successional forbs, grasses, shrubs, and trees (Confer 1992). Prior to human habitation in North America, this type of cover was created by wildfires, wind storms, large grazing herbivores and beavers (Confer 1992, Hunter et al. 2001, Lorimer 2001). Although the size of the Golden-winged population prior to European colonization is unknown, the population may have peaked in early 20th century because of the creation of early successional vegetation resulting from the abandonment of farm land (Confer 1992, Lorimer 2001). More recently, reforestation of abandoned farm land, reduced timber harvest, and fire suppression have contributed to the loss of early-successional habitat in the eastern United States (Hunter et al. 2001, Lorimer 2001).

Early successional stages of plant communities are ephemeral, and species occupying these areas often use a range of successional stages. Indeed, Golden-wingeds occupy a variety of vegetation types including tamarack (*Larix* spp.) swamps (Will 1986), aspen (*Populus* spp.) stands (Roth and Lutz 2004), utility rights-of-way (Kubel and Yahner 2008), abandoned farmland and agricultural fields (Will 1986, Confer 1992, Demmons 2000, Reed et al. 2007), montane wetlands (Rossell et al. 2003), and regenerating forests with herbaceous ground cover and scattered openings (Klaus and Buehler 2001, Confer et al. 2003, Vallender et al. 2007, Kubel and Yahner 2008). Golden-wingeds also inhabit reclaimed coal mine land in Pennsylvania, Maryland, West Virginia, Virginia, Kentucky, and Tennessee, where contour and mountain-top removal mining have created linear openings and larger expanses of grasslands and/or shrublands in an otherwise forested landscape (Canterbury et al. 1996, Patton et al. 2004, Bulluck and Buehler 2008).

Although Golden-wingeds are known to occupy reclaimed strip mines throughout the Cumberland Mountains of eastern Tennessee, it is unknown how long it takes these sites to succeed into later seral stages and no longer provide nesting cover. It also is unknown when nest success is maximized along this successional gradient. Furthermore, it is unknown to what extent management, such as prescribed fire, can restore and maintain suitable nesting cover on reclaimed coal mine lands in this region.

In the southern portion of their breeding range, Golden-winged populations are limited to higher elevations in the Cumberland Mountains (>480 m; Welton 2003, Patton et al. 2010) and Southern Blue Ridge (>730 m; Klaus and Buehler 2001). Previous field research in this region found Golden-wingeds occupied forest stands (age ≤13 yr) that had lower basal area (median = 10 m^2/ha) and lower sapling densities (median = 4,445 stems/ha) than unoccupied stands located in regenerating northern hardwood and mesic oak-hickory forests (Klaus and Buehler 2001). Klaus and Buehler (2001) also compared song perch sites to nest sites and found sapling densities were lower (median = 889 stems/ha) and tree canopy cover was less (38.8%) at nest sites. On reclaimed coal mines in southeastern Kentucky, Patton et al. (2010) found Golden-winged territories had greater grass and canopy cover when compared with Blue-wingeds, and all observed Golden-winged territories included an edge between reclaimed mines and mature forest. Other studies have also reported an association with edge throughout the breeding range (Confer et al. 2003, Rossell et al. 2003, Roth and Lutz 2004, Martin et al. 2007, Kubel and Yahner 2008).

More specific to our study sites in the Cumberland Mountains, Bulluck and Buehler (2008) found percent cover of herbaceous vegetation (i.e., grasses and forbs) was greater at nest sites than non-nest sites, and percent cover of woody vegetation was greater at non-nest sites. Similar to Klaus and Buehler (2001), Bulluck and Buehler (2008) found percent cover of saplings was less at nest sites than non-nest sites.

Based on previous research, I hypothesized that territory selection and nest-site selection were not random, but that male and female Golden-wingeds select specific habitat features relative to availability. I predicted territory selection and nest-site selection would be positively influenced by herbaceous ground layer, shrub cover and proximity to forest edge. To test these hypotheses, I compared habitat variables within male Golden-winged territories to those in unoccupied sites, and I compared micro-habitat variables measured within plots centered on nest-sites to non-nest plots located within Golden-winged territories.

Additionally, I hypothesized the decline in Golden-winged abundance on our study sites were related to an increase in later successional stages. To test this hypothesis, I compared habitat variables measured within used nest-site plots to variables measured within plots where Golden-wingeds once nested, but no longer inhabit. I predicted an increase in canopy cover has resulted in a decrease in herbaceous ground cover necessary for nesting.

Lastly, I evaluated the relationship between habitat variables, prescribed burning treatments, and daily nest survival. Habitat may influence breeding productivity, but selection of specific habitat features may not always be linked to nest survival (Dinsmore et al. 2002, Boves 2011). Field studies have found Golden-wingeds can be attracted to sink areas, where recruitment does not sustain the population (Kubel and Yahner 2008). Thus, management plans should not be based on presence-absence data alone (Jones 2001, Morrison 2001). Understanding the effects of habitat variables and management techniques on Golden-winged nest survival is critical to the conservation of this declining species. Fire can directly alter composition and densities within predator assemblages (Lyon et al. 2000). Fire also alters composition and/or structure of vegetation, which may affect nest concealment and therefore predation rates. I hypothesized prescribed fire would influence Golden-winged nest success. Because burning may initially thin vegetation, I predicted nests located in areas which were burned within ≤1 year of nesting would have lower daily survival rates than nests located in areas which were burned ≥2 years prior to nesting attempts.

Methods

Study area

My study sites consisted of five mountaintop sites of both managed (e.g., prescribed fire, timber harvest, vegetation thinning, grass and forb seeding) and unmanaged land located within and adjacent to the North Cumberland Wildlife Management Area (Fig. 1.1). Two managed sites, Ash Log and Massengale Mountains, were selected for nest searching in 2009-2011 because they had known concentrations of breeding Golden-wingeds (Bulluck 2007). Three unmanaged sites, Red Oak, Fork, and Burge Mountains, were selected for re-sampling nest-site habitat variables that were first measured during 2005 and 2006 because the Golden-winged population had shown a decline since that time period (see Chapter 1). Mean elevation of all five sites was 850 m (range = 770 – 960 m).

The predominant land cover of the region was a combination of mixed-mesophytic and oak- hickory forests (Wade et al. 2000, Artman et al. 2005). Approximately 15% of the region was in early stages of succession because of timber harvest and surface mining for coal (Bulluck 2007). On my study sites, coal surface-mine reclamation occurred from about 1980–1990 (Bulluck and Buehler 2008). Reclamation involved planting black locust (*Robinia pseudoacacia*), tall fescue (*Schedonorus phoenix*), sericea lespedeza *(Lespedeza cuneata)*, and autumn olive (*Elaeagnus umbellata*). Yellow-poplar (*Liriodendron tulipifera*), maples (*Acer* spp.), oaks (*Quercus* spp.), blackberry (*Rubus* spp.), and a variety of forbs (e.g., *Solidago* spp., *Aster* spp.) and grasses (e.g., orchard grass, *Dactylis glomerata*; timothy, *Phleum pratense*) have since colonized the sites.

Currently, Massengale Mountain is the only site under experimental management for Golden-wingeds. Three burn units, measuring 40 ha (zone 1), 115 ha (zone 2), and 145 ha (zone 3), were on a one- to three-year burning rotation (Figure 2.1). A single unit, measuring 35 ha, was burned on Ash Log Mountain in 2007, but logistical constraints prevented subsequent prescribed burning. Although prescribed burning as an experimental treatment is difficult to standardize, all burns were conducted during the dormant season. Prescribed burns were of low-moderate intensity with flame heights generally 1-2 m. The objective of these prescribed burns was to promote herbaceous vegetation and reduce woody vegetation. Fire-return intervals varied on Massengale Mountain based on effectiveness of the initial burn treatment. Zone 1 was burned in 2008, 2009, and 2011; zone 2 was burned in 2007 and again in 2010; and zone 3 was burned in 2008 and 2011 (Figure 2.1). Damp and/or green vegetation, as well as minimal fuel layers in some areas, contributed to less effective burns in some years.

Field methods

I began monitoring the arrival of Golden-wingeds mid-April, 2009-2011. I opportunistically searched for nests throughout territories following the guidelines of Martin and Geupel (1993). I took precautions during nest searching and monitoring to minimize observer effects on nest survival (e.g., cueing predators to nests). Once found, I re-visited nests every 1 to 4 days to determine fate (i.e., survival or failure). Nests were considered successful if ≥1 Golden-winged young fledged.

Territory data collection

Behavior of individual males (e.g. movements, interactions with neighboring males, song perches) that indicated territory boundaries was noted while nest searching. I used mist-nets to capture individuals and band them with unique combinations of ≤3 color bands (Avinet, Inc.) and one USFWS uniquely numbered aluminum band, which allowed me to subsequently identify individual males and distinguish among overlapping territories. Occasional song and/or phenotypic variations also allowed me to identify some individuals. Approximately 70% of the territorial males were color-marked each year.

I sampled vegetation throughout the territory of the male for each nest that was located using a systematic sampling design (Figure 2.2). Thirty 1-m^2 sampling locations were located along line transects. To locate transects, I generated sets of random starting locations, compass bearings, and distances (m) between sampling points along each transect before starting field work. Garmin GPS units were used to record start and end locations for each transect, as well as distance measurements between transect lines and sampling locations. Transects were parallel

≥25 m apart to ensure plots did not overlap. Sampling points and transects were distributed across each territory to cover the area of the territory and conform to the shape of the territory.

At each sampling location on a given transect, percent cover of grasses, forbs, *Rubus* spp., vines, course woody debris, litter, and bare ground were visually estimated, summing to 100% cover. Above the ground layer, percent cover of shrubs <1 m in height, shrubs >1 m in height, saplings <10 cm dbh, and canopy cover contributed by trees >10 cm dbh were visually estimated. Distance (m) to microedge (defined as an obvious change in vegetation height or composition), was also visually estimated at each of the 30 sampling locations in each territory.

At every 5th sampling location, additional vegetation parameters were recorded inside 5-m and 11.3-m radius plots. Within the 5-m plots, I counted the number of shrubs 1-2 m tall, the number of shrubs >2 m tall, and the number of saplings 1–10 cm dbh and >0.5 m tall. Within the 11.3-m plots, I counted the number of snags >10 cm dbh and determined basal area using a 2.5 m^2/ha prism.

To determine male territory selection, unoccupied habitat was sampled by extending transect lines beyond the territorial boundary, avoiding entry into territories occupied by adjacent conspecifics. If all adjacent area was occupied, transects were placed in non-adjacent yet apparently suitable areas (i.e., areas that contained grass, forb, shrub, and some canopy cover; not including food plots that completely lacked shrub and canopy cover). I tested the null hypothesis that habitat comprising Golden-winged Warbler territories does not differ from unused habitat.

Nest-site data collection

I used a nested plot design (1-m and 11.3-m radius plots) to sample habitat features at each nest site and a single paired random site, to test the null hypothesis that habitat at nest sites does not differ from available habitat located inside the male's territory. Random sites were 25 to 50 m from the center of the nest site and located inside male Golden-wingeds defended territory and therefore, were available for use (Figure 2.3; Morrison 2001). For each nest site, I measured nest height, recorded the plant species the nest was built in, and the height of plant(s) in which the nest was built. At plot center (for both nest and random sites), I visually estimated percent covers of forbs, grasses, *Rubus* spp., additional woody vegetation (e.g., tree sprouts), litter, vines, and bare ground within 1 m^2. Litter depth (cm) was measured 1 m from plot center in each cardinal direction. Within the 11.3-m plot, percent cover estimates were obtained using an ocular tube (James and Shugart 1970). Percent cover estimates of canopy trees (>10 cm dbh), small saplings (<1 m in height), large saplings (>1 m in height), shrubs, *Rubus* spp., forbs, grasses, and vines were generated by taking 20 ocular tube measurements at 5 points along each of four 11.3-m transects radiating from plot center in each cardinal direction (percent cover = # readings with cover type/20 x 100%). At each sampling point, an observer looked through the ocular tube straight up at the canopy and down at the ground (held at 45 degrees from body) and recorded the vegetation classes within the ocular tube view. We also recorded the three most dominant plant species within each vegetation class based on the number observed with the ocular tube.

Similar to Nudds' (1977) "vegetation profile board," we measured visual obstruction from vegetation around each nest and random site using a 2-m tall density board containing 2 columns of ten 20-cm x 20-cm cells. One person stood with the board at plot center while the observer viewed it from 10 m away in each cardinal direction. We recorded the number of cells that were >50% covered with vegetation. Percent cover equaled the number of cells covered/20 x 100%.

Within 11.3 m (0.04 ha) of plot center, we record the number of snags (>10 cm dbh), average shrub height (m), average sapling height (m), and visually estimated to the nearest 10 m the distance from plot center to a mature forest edge. A prism (2.5 m^2/ha) was used to determine basal area.

Re-measurements at old nest sites

Abundance data showed a positive correlation between the Golden-winged Warbler population and prescribed burns on Massengale Mountain (Fig. 1.2). Conversely, population decline may have occurred on three study sites (Red Oak, Fork and Burge) where no management action had been implemented (Fig. 1.2). During 2010, we returned to a random sample of nest-plot sites where habitat measurements were collected during 2005 and 2006, and re-sampled the same habitat variables to document successional changes. Data collection followed the nested-plot design described above.

Statistical analysis

Statistical analysis for territory selection, nest-site selection, and differences relative to previous nest sites were all performed using SAS statistical software version 9.3 (SAS Institute 2011). Measurements were averaged across the 30 sampling locations for each territory, which provided a representative mean with a small confidence interval for the highly patchy distribution of vegetation that is typical for Golden-wings.

Vegetation data were collected for every nest we monitored, which included several pairs of either male and/or female Blue-winged/Golden-winged hybrids. The nest-pairs that included either a male or female Blue-winged were excluded from nest vegetation analysis (n = 4), as were the territory vegetation data collected in male Blue-wingeds' territories (n = 2). I pooled the hybrid and Golden-winged datasets, however, after testing for significant differences and finding none.

Several habitat variables were not normally distributed and/or had unequal variance (i.e., heteroscedasticity); namely, random and unused plots had greater variance than nest and used plots, respectively. Thus, the NPAR1WAY procedure was used for nonparametric tests of differences, based on Wilcoxon scores, for our two-sample datasets (SAS Institute 2011). All statistical tests were two-tailed. As the criterion for statistical significance, the Dunn-Sidak method was used to correct the α-level, which is

$$1 - (1 - \propto)^{\frac{1}{n}}$$

where n is the number of independent tests. This method controls Type I error (i.e., falsely rejecting the null hypothesis; Gotelli and Ellison 2004). This correction is conservative, and so regardless of the adjusted α, I considered relationships with P-values ≤ 0.05 potentially biologically relevant and deserving of consideration for subsequent modeling of nest success (Gotelli and Ellison 2004).

Nests placed in locations similar to previous years were considered independent because annual vegetation changes occur (e.g., growth and/or disturbances). Thus, nesting data were pooled across years. Habitat selection data from my two nest-searched sites (Ash Log and Massengale) were analyzed separately for each site because statistically significant differences between sites for several habitat measurements were detected. Furthermore, Massengale had received annual prescribed burning treatments on one of three burn-units since 2007, for a total of seven prescribed burns, whereas Ash Log received a single burn treatment in 2007 within a single burn-unit.

The SAS CORR Procedure was used to identify correlations among continuous habitat variables (SAS Institute 2011). Because several variables displayed non-normal distributions, Spearman rank-order correlation coefficient (r > 0.75), a nonparametric measure of correlation, was used for identifying collinearity among variables (SAS Institute 2011).

Modeling nest survival

I used the nest survival module in Program MARK to obtain estimates of daily nest survival rates (DSR) (Rotella et al. 2004, Rotella 2010). The logistic-exposure method is used in Program MARK to estimate the probability that a nest survives a single day. This approach is an improvement over the Mayfield estimator because it does not require constant DSR for all nests in a sample and models can contain categorical and continuous variables representative of individual nests (Dinsmore et al. 2002, Rotella et al. 2004). Using the logit link function, constraining survival to the binomial interval (0, 1), DSR of a nest on day *i* is modeled as

$$DSR = \frac{exp(\beta_o + \Sigma_j \beta_j x_{ji})}{1 + exp(\beta_o + \Sigma_j \beta_j x_{ji})}$$

where β_j are coefficients to be estimated and x_{ji} are values for *j* covariates on day *i* (Rotella et al. 2004). DSR can then vary among groups of nests, among individual nests, and among days (Rotella 2010). Parameters (*βj*) of each model are estimated iteratively (Rotella 2010). True probability of a nest surviving from initiation to completion ("nest survival") is then calculated as $(DSR)^n$, where *n*, the total length of the nesting period, equaled 25 days for Golden-wingeds on our study site (5 days laying + 11 days incubating + 9 days brooding).

Candidate models were evaluated using an information-theoretic model selection approach (Burnham and Anderson 2002). I also chose a hierarchical approach to model selection because the addition of models to the results browser influences model weights (Rotella 2010). Effective sample size, i.e. exposure days, were used to compute Akaike's information criterion (AICc), which incorporates a correction for small sample size. Models were ranked relative to the model with minimum AICc value, and the relative distances (ΔAICc) between the best model and each competing model (Rotella 2010). Normalized Akaike weights (w_i) are estimates of the probability of model *i* being the best model given the data and the model set (Anderson 2007). A constant (intercept only) survival model is equivalent to the Mayfield estimate, and was included in all model sets for comparison.

In the modeling hierarchy, the first suit of models explored the additive and interactive effects that year, seasonal time trends, nest age, and weather may have on Golden-winged (and hybrid) daily nest survival rates. Specifically, I modeled variations in the DSR as a function of 1) the categorical effect of year (2009, 2010 and 2011), 2) linear time trend effects within the breeding season, 3) quadratic time trend effects within season, 4) linear effect of nest age, 5) quadratic effect of nest age, 6) cubic effect of nest age, and 7) three weather covariates - maximum and minimum daily temperature and daily precipitation. I obtained temperature and precipitation data from the nearest National Oceanic and Atmospheric Administration (NOAA) weather station, located in Oak Ridge, TN (ID: GHCND:USW00053868).

As many authors have noted, variability in daily nest survival rates are commonly correlated with temporal changes (Dinsmore et al. 2002, Jehle et al. 2004, Rotella et al. 2004, Grant et al. 2005). Because fluctuations in predator communities and/or weather patterns may occur annually and/or seasonally, I hypothesized that annual variation in nest survival rates would occur. Similarly, I hypothesized that daily nest survival would vary across the nesting season. Golden-wingeds re-nest following nest failure, in which case a bimodal pattern (i.e., quadratic time trend) may best reflect nest survival rates. Daily survival rates may also be closely associated with changes in nest age and/or stage (e.g. laying, incubating, nestling) as parental activity increases to feed begging young, subsequently cueing predators to nest location. In this case, a cubic pattern for nest age may best reflect nest survival rates. Specific to the effect of weather, temperature and/or precipitation may be the ultimate cause of nest failure or, again, predator activity may respond to changing weather patterns. Hence, I hypothesized that extreme daily temperatures and/or precipitation events would reduce daily nest survival rates.

In the second suite of models in the hierarchy, I investigated the primary question of this study; are Golden-winged daily nest survival rates related to the burn history of the nesting habitat? Because burning may thin vegetation, at least initially, I hypothesized that nests located in areas that were burned within one year of nesting would have lower DSR than nests located in areas that were burned ≥2 years prior to nesting attempts. In anticipation of each burn classification having relatively small sample sizes, I chose to investigate a site-level effect (i.e., Ash Log vs. Massengale) on nest success within the same suite of models. Massengale Mountain has been burned annually since 2007, for a total of seven prescribed fire treatments, whereas Ash Log was burned a single time during 2007.

In a third suite of models in the hierarchy, I investigated the relationship of habitat variables measured within the nested-plots to DSR of nests. Habitat management can manipulate nesting vegetation composition and structure. Fire, for example, kills some vegetation while stimulating the seed bank and increasing the growth of other species (e.g., *S. lespedeza*; Harper 2007). Minimal effects of vegetation structure and composition on Golden-winged nesting success have been identified, though (Confer et al. 2003, Bulluck and Buehler 2008). Vegetation may also have varying degrees of influence relative to nest proximity (Grant et al. 2006); therefore, habitat variables were considered at multiple spatial scales. In this suite of models, I investigated the relationship of percent cover to DSR for the primary vegetation types within 11.3 m of the nest: 1) *Solidago* spp., 2) *S. lespedeza*, 3) total forbs (which include *Solidago* spp. and *S. Lespedeza*), 4) grasses, 5) vines, 6) shrubs, 7) *Rubus* spp., 8) saplings above waist height,

9) saplings below waist height, and 10) canopy. Similarly, I investigated the relationship of percent cover to DSR for vegetation within 1 m of the nest: 1) total forbs, 2) grasses, 3) vines, 4) *Rubus* spp. At the finest scale, I investigated the relationship of the presence of specific vegetation in the nesting substrate to DSR including 1) *Solidago* spp., 2) *S. Lespedeza,* 3) grasses, 4) *Rubus* spp., and 5) tree saplings. In addition to percent cover estimates and presence in nest substrate, I investigated the relationship of 1) distance (m) to the nearest forest edge, and 2) mean vegetation density from cover board estimates to DSR. In total, 21 habitat covariates were modeled for their relationship with daily nest survival rates.

I decided a priori that a final suite of models would incorporate the top models from each preceding suite, if a given model performed better than the constant survival model (i.e., intercept only), and/or had a ΔAICc ≤ approximately 4 (Anderson 2007). To address issues of sample size, I included the nests of male and/or female hybrids in the dataset for modeling sources of variation in DSR. I did, however, remove the nests of Blue-wingeds from the dataset because differences in breeding habitat between the species have been documented on other study sites (Confer and Knapp 1981, Confer et al. 2003, Patton et al. 2010).

Results

Territory selection

We collected vegetation within 77 used-territory plots and 49 unused-territory plots. Sample sizes differed because the availability of unused plots was limited, as most territories were adjacent to the territorial boundaries of neighboring males. Although there were differences between habitat variables between used and unused plots for our two study sites, the structural composition within used territory plots was similar between sites. Within used territory plots, the most abundant groundcover included forbs (median = 44.3%, range = 11.8 – 76.0%), grasses (median = 18.4%, range = 1.83 – 53.2%) and *Rubus* spp. (median = 11.4%, range = 5.43 – 48.9%). Saplings were the most abundant midstory cover (median = 16.2%, range 0 – 44.7), and median tree canopy cover was 8.7% (range 0 – 44.7%) (Table 2.1).

Of the 11 habitat variables tested for differences between used and unused territory plots, none differed based on the Dunn-Sidak adjusted $\alpha \leq 0.00465$ for both study sites. On Massengale, mean percent groundcover of forbs was greater within unused plots than used plots ($P = 0.010$) and percent ground cover by grasses was greater in unused plots than used plots ($P = 0.012$). On Ash Log, *Rubus* spp. was more abundant within used plots than unused plots ($P = 0.044$) (Table 2.2). No other groundcover differed between used and unused territory plots on either Massengale or Ash Log ($P > 0.05$, Table 2.2). Total mid-story cover by shrubs was 9.61 and 5.62% for Massengale and Ash Log, respectively, and shrubs greater than 1 m in height were more abundant within used plots than unused plots ($P < 0.001$) on Massengale only. The most commonly recorded shrub species were autumn olive (*Elaeagnus umbellata*), elderberry (*Sambucus canadensis*), sumac (*Rhus copallinum* and *Rhus glabra*), multiflora rose (*Rosa multiflora*) and *Lespedeza bicolor*. Midstory cover by saplings was ~18% for both sites; sapling cover was greater in used plots than unused plots ($P = 0.038$) on Massengale (Table 2.2).
The number of saplings within 5-m radius plots with a DBH <10 cm was 12.6 and 17.8 for Massengale and Ash Log, respectively, and neither site differed between used and unused territory plots ($P > 0.05$). Percent tree canopy cover was 9.8 and 14.4% within used territory plots on Ash Log and Massengale, respectively; canopy cover was greater in unused territory plots only on Massengale (31.2%, $P = 0.002$). Similarly, basal area was less on used territory plots than unused plots for both Ash Log (2.6 and 7.89 m^2/ha, respectively; $P = 0.001$) and Massengale (6.4 and 12.1 m^2/ha, respectively; $P = 0.007$). The three most common trees on both sites, within used and unused plots, were black locust (*Robinia pseudoacacia*), maples

(*Acer* spp.), and yellow-poplar (*Liriodendron tulipifera*). Distance to microedge was 2.6 m and 2.8 m for Massengale and Ash Log, respectively. Distance to microedge on Massengale was greater within unused territory plots (5.6 m, P = 0.003).

Nest-site selection

Of the variables collected within the nested plots, percent canopy cover and basal area showed some correlation (r = 0.56253, P < 0.0001). I retained both variables in the analysis because the correlation was not strong, and canopy cover included cover by saplings not included in the basal area count. No other variables were correlated within the nested plots, thus 14 variables were tested for differences between nest sites and non-nest sites (Table 2.3).

No nested plot variables differed between nest and non-nest sites based on the Dunn-Sidak adjusted α-value = 0.0037 (Table 2.3). Overall structure and composition within nest plots was largely similar between the two sites, but differences were notable for a few variables. Percent cover by shrubs within 11.3 m of plot center was greater on Massengale than Ash Log (13.5% cover and 1.4%, respectively; P < 0.0001). The most common shrubs on Massengale were autumn olive, multiflora rose, *Lespedeza bicolor,* and elderberry. The number of snags was greater on Massengale than Ash Log (0.83 and 0.21, respectively; P = 0.001).

Within 1 m of nest sites, forb cover was greater in the nest plot than non-nest plot on Massengale (P = 0.05), and woody cover was greater within nest plots than non-nest plots on Ash Log (P = 0.019). Within the 11.3-m plot, density board measurements were greater within nest plots then non-nest plots on Ash Log (P = 0.007), as was sapling cover <1 m (P = 0.026) and canopy cover (P = 0.038). On Massengale, only shrub cover at the 11.3-m scale differed between nest plots and non-nest plots (13.5% and 10.2%, respectively; P = 0.044; Table 2.3).

Re-measurements at old nest sites

Cover estimates of saplings >1 m decreased from 26.9% in 2005/2006 to 8.0% in 2010 (P = 0.002) and cover estimates of saplings <1 m decreased from 11.4% to 1.0% (P = 0.0006; Table 2.4). In conjunction with those two variables, average sapling height increased from 1.8 m to 3.2 m (P = 0.0001). The increase in canopy cover, however, was less apparent (50.5% to 64.6%, P = 0.0556). Vine cover increased from 22.7% to 44.5% (P = 0.02), with *Clematis* spp. being the most abundant vine. *Rubus* spp. was not independently measured from all other shrubs during 2005/2006, as it was during 2010. An approximate comparison can be made by adding shrub and *Rubus* spp. cover estimates together from 2010, which showed a decrease over time from 43.1% in 2005/2006 to 15.0% in 2010 (P < 0.0001; Table 2.4).

Nest Survival

Of 19 models in the first model suite designed to explain daily nest survival rates, all top models included the categorical effect of year. The model with the most support included the quadratic effect of time in addition to the effect of year (w_i = 0.454), which performed well above the model that assumed constant survival (ΔAICc = 16.13; Tables 2.5 and 2.6). Parameter estimates from the best model showed that annual variation in nesting success was pronounced, ranging from a low of 10.8 ± 5.4% in 2009, to a high of 57.5 ± 8.8% in 2010 (Table 1.6). The quadratic time trend effect within the breeding season showed survival followed a curvilinear pattern, where survival was greatest in early May, declined through the rest of May and into early June, and increased from approximately the second week in June through the remainder of the nesting season (Figure 2.4). There was marginal support for the second-ranked model that included the effect of year, quadratic time, and maximum daily temperature (ΔAICc = 2.01, w_i = 0.166; Table 2.6), but the 95% confidence intervals for the maximum daily temperature β estimate included zero ($-0.062 \leq \beta_{\text{MaxTemp}} \leq 0.055$). A global model that included the effect of year, quadratic time, quadratic age, and maximum daily temperature (variables that individually ranked above constant survival) did not have much support (ΔAICc = 4.49, w_i = 0.07).

In the second model suite, burn history was included as an individual covariate for each nest: Year of burn (n = 4 nests) and 1 year post-burn (n = 8 nests; total n for group = 12); 2 years post burn (n = 4 nests), 3 years post burn (n = 4 nests), and 4 years post burn (n = 4 nests; total for group = 12); and control, i.e. no burn (n = 68). Site was also included as an individual covariate: Ash Log (n = 68 nests) and Massengale (n = 24). Neither the burn history nor site model performed better than the model that assumed constant survival (Table 2.7). DSR parameter estimates were similar between sites and among burn history groups (Table 2.8). Although the model that included site had a ΔAICc value of 2, the 95% confidence intervals for the β estimate spanned zero ($-0.693 \leq \beta_{\text{site}} \leq 0.652$).

Within the suite of models that included only habitat variables, four models ranked above the model assuming constant survival (Table 2.9). The model with the most support included the presence of *Rubus* spp. in the nest substrate (ΔAICc = 0, AICc weight = 0.237). Furthermore, the 95% confidence interval for the β estimate for *Rubus* spp. in the nest substrate was the only one that did not span zero ($-1.426 < \beta_{\textit{Rubus}\text{ spp. in nest substrate}} < -0.152$; Table 2.10). The model which included percent cover of *Rubus* spp. within 1 m of the nest had more limited support (ΔAICc = 2.19, AICc weight = 0.091; Table 2.9). The model which included distance to the nearest forest edge had marginal support (ΔAICc = 2.82, AICc weight = 0.079; Table 2.9), as did the model which included percent cover of *Rubus* spp. within 11.3 m of the nest (ΔAIC = 3.24, AICc weight = 0.058; Table 2.9). Nest survival decreased with the presence of *Rubus* spp. in nest substrate and increasing percent cover at the 1-m and 11.3-m plot scales (Table 2.10 and Figure 2.5a-c). Nest survival also decreased with an increase in distance from the forest edge (Table

2.10 and Figure 2.5d). Average forest edge distance was 25.9 m (range = 0 – 200 m). All other habitat variables were ranked below the model assuming constant survival with ΔAIC value >3.5.

Within the final suite of 16 models (Table 2.11), the most supported models all included the year and quadratic time effect. The top ranking model also included *Rubus* spp. within the nest substrate (ΔAIC = 0, AICc weight = 0.271). The second ranked model included *Rubus* spp. within 1 m of the nest (ΔAIC = 0.01, AICc weight = 0.269). The third ranked model included two habitat variables – the additive effect of *Rubus* spp. at the finest scale (within nest substrate), where its effect on nest success was strongest, and the nearest distance to a forested edge (ΔAICc = 1.70, AICc weight = 0.115). Although there was some support for this model, again the 95% confidence interval for the forested edge parameter estimate included zero ($-0.013 \leq \beta_{\text{forest edge}} \leq 0.007$).

Discussion

Territory selection

Strong territory selection relative to availability was not evident when sites were analyzed independently. Vegetation structure and composition within used territory plots was similar between Ash Log and Massengale. Selection for shrubs >1 m in height on Massengale, relative to Ash Log, was perhaps a response to the lesser abundance of *Rubus* spp. Indeed, the sum of *Rubus* cover and shrub cover was ~20% for both sites, which is consistent with percent shrub cover estimates reported elsewhere.

Unlike regenerating clearcuts, herbaceous ground cover was not a limiting factor to Golden-winged breeding habitat on the reclaimed coal mines we studied. Klaus and Buehler (2001) documented lower sapling densities in occupied territories than unoccupied, but on

Massengale sapling density was much less than within regenerating clearcuts. On Massengale, Golden-winged habitat often graded into adjacent mature forest. As a result, unused territory transects frequently entered into mature forest. Thus, unused plots had greater basal area and canopy cover than used plots. Although forbs and grasses had greater abundance on unused than used plots on Massengale, this result may be misleading because ground cover was composed of shade-tolerant forbs and grasses (e.g., stilt grass) that were not used as nesting substrate. Moreover, the same degree of certainty for used territories cannot be applied to unused territories because Golden-wingeds likely foraged well into mature forest, without displaying territorial defenses.

Nest-site selection

Nest-site selection was similar to territory selection in that significant differences ($P < 0.05$) between variables recorded within nest plots and non-nest plots differed between Ash Log and Massengale. Shrub cover other than *Rubus* spp. was greater on Massengale. The sum of *Rubus* spp. and shrub cover was ~50% for both Massengale and Ash Log, again highlighting the similarity in structure of vegetation across both sites. Bulluck (2008) found differences between nest and non-nest plots for four vegetation variables recorded from 2004-2006: forb, grass, and woody cover within 1 m, and percent sapling cover within 11.3 m of nest sites. None of these variables differed ($P < 0.05$) between nest and non-nest plots during 2009-2011. Nest-site selection at these scales did not show a consistent pattern across years on our study sites within the same management area.

Re-measurements at old nest sites

Comparing cover estimates between older (2005-06) and more recent (2009-2011) time periods, forb cover was approximately 10% greater, grass cover was approximately 35% less, and woody cover was approximately 16% less within 1 m during 2009-2011. Percent cover measurements at the 1-m scale were visual estimates; these changes may be indicative of true vegetation changes, site differences, or difference among observers. Percent sapling cover within 11.3 m was approximately 20% less in nest plots and 33% less in non-nest plots during 2009-2011, compared to 2005-2006. Percent cover measurements at the 11.3-m scale were estimated more objectively using an ocular tube; the differences more likely reflect true vegetation changes.

Because ocular tube readings for percent cover measurements may be more objective, I compared re-measurements recorded at the sub-sampled 2005-2006 plots at the 11.3-m scale only. As predicted, succession has occurred on older reclaimed coal mines. The increase in sapling height may explain the decrease in sapling cover recorded with ocular tube readings, as those are taken with the field of view directed towards the ground at a 45-degree angle. Most notable was the change in shrub and/or *Rubus* spp. cover. Shrub and *Rubus* spp. cover combined showed a significant decrease over time from 2005-06 to 2010-11. Canopy cover, however, showed only a slight (10%) increase over this time period, possibly because it takes longer for tree growth to change canopy cover conditions. *Rubus* spp. occurs in open to semi-shady and densely-shaded forests (Miller and Miller 2005). Without a pronounced increase in canopy cover, I cannot explain the marked decrease in shrub cover over time. Nonetheless, it would appear that the population decline on these sites is correlated with a loss of shrub cover and/or increase in vine cover.

Nest survival

I found stronger support for annual and seasonal variability in Golden-winged daily nest survival rates than previously documented by Bulluck and Buehler (2008) within the same wildlife management area. Variability in daily nest survival rates within the nesting season, and the more pronounced variability among years that I documented may reflect temporal changes within the predator community (e.g., density and species). The lack of specific nest predator data permits only speculation about the ultimate causes of nest failure. Potential nest predators seen on our study sites included, but were not limited to, eastern garter snake (*Thamnophis sirtalis sartalis*), black rat snake (*Elaphe obsoleta*), timber rattlesnake (*Crotalus horridus*), copperhead (*Agkistrodon contortrix*), eastern chipmunk (*Tamias striatus*), eastern gray squirrel (*Sciurus carolinensis*), coyote (*Canis latrans*), and northern raccoon (*Procyon lotor*). Annual variation in nest survival may correspond to temporal pulses of small mammal (e.g. eastern chipmunk and grey squirrel) abundance, known to occur in conjunction with mast production (McShea 2000, Schmidt 2003, Schmidt and Ostfeld 2003). Decreased nest survival rates through the middle of the nesting season may also reflect a change in predator activity, e.g., increased snake activity with the rise in temperature (Weatherhead et al. 2010). Increased nest success late in the season may have occurred because vegetation growth increased nest concealment (Grant et al. 2005).

Golden-wingeds display strong selection for mixed scrub-shrub vegetation across their breeding range, and this may come at a reproductive cost on reclaimed coal mining land where woody encroachment by *Rubus* spp. was associated with increased nest predation.
Bulluck and Buehler (2008) found some evidence that Golden-winged nest survival decreased with a woody stem in the nest substrate. Similarly, I found a negative relationship between DSR and *Rubus* spp. in nesting substrate. Thus, I concluded micro-scale habitat characteristics did appear to affect daily nest survival, and therefore predation rates. The decrease in nest survival with the presence of *Rubus* spp. in the nesting substrate suggests that *Rubus* spp. may increase nest detectability to predators. Alternatively, *Rubus* spp. may also provide protective cover for

potential nest predators, thereby increasing incidental nest predation. Large annual variability in nest survival, as well as within breeding season, may prevent fine-scale adjustment of breeding strategies or nest-site selection to predation risk. Alternatively, Golden-winged females may be selecting *Rubus* spp. for nesting to provide easy access to protective cover during the vulnerable post-fledging period, although at a cost in nest success.

Our study sites within the Cumberland Mountains of Tennessee are near the southern limits of the Golden-winged breeding range. Within this region, they are primarily restricted to the highest elevations. Despite indication that global climate change may be causing a general northward range shift, this relationship with overall population decline is not clear (Buehler et al. 2007). The Cumberland population has remained stable on several study sites, 2003-2011, and increased where prescribed burning is active. Moreover, fire has not caused decreased nesting success. However, the effects of fire on predator habitat quality and subsequent nest predation needs further study. Despite their relative frequency, predation events were never directly observed. Thus, we are currently limited by our inability to definitively identify the relative roles of individual nest predators in Golden-winged Warbler demography. These results highlight the need to identify specific Golden-winged nest predators and understand factors that affect their distribution and abundance on coal mine land in the Cumberland Mountains. Understanding the predator community and its response to habitat management for the Golden-winged Warbler may elucidate the ultimate causes, and perhaps pattern, of nest failure. Research should continue to monitor the response of Golden-wingeds to prescribed burning, ultimately detailing a regime that maintains suitable nesting habitat.

Management Implications

Reclaimed coal mine lands are appealing for conservation of Golden-winged Warblers because the disturbance creates openings of early successional plant communities in an otherwise forested landscape (Canterbury et al. 1996, Patton et al. 2004, Bulluck and Buehler 2008). Golden-wingeds did not nest, however, in areas that were reclaimed solely with herbaceous vegetation. Habitat management for Golden-wingeds must include some tree and shrub cover as well. Based on a study of Golden-winged and Blue-winged habitat on reclaimed coal mining land in Kentucky, Patton et al. (2010) recommend scattered trees (basal area = 2-8 m^2/ha), ~20% cover by shrubs, and ~40% cover by grasses and forbs.

The Cumberland Mountains of eastern Tennessee contain about 75,000 ha of state-owned contiguous forest. Without the constraints of private ownership, the North Cumberland Wildlife Management Area has great potential for Golden-winged Warbler habitat management. Prescribed burning is a viable management practice for restoring and/or maintaining early successional habitat for disturbance-dependent wildlife species (Hunter et al. 2001, Harper 2007, Klaus et al. 2010). Not all burns are the same, however; timing and frequency of burning influences vegetation responses (Miller 2000, Harper 2007). Specifically, late dormant season burns in early successional plant communities stimulates warm-season grasses and forbs (Harper 2007). Growing season fires may better control woody encroachment and maintain openings in woodlands (Wade et al. 2000, Harper 2007). Thus, managers will have to consider site-specific vegetation composition before implementing prescribed burns.

Massengale is larger than Ash Log (162 ha and 125 ha, respectively). Yet the number of territorial males remains lower on Massengale than Ash Log (24 males and 38 males, respectively, 2011). Herbaceous cover and total shrub cover are approximately the same across both sites. They differ, however, in the amount of cover by saplings and mature trees. The number of saplings (DBH <10 cm) is lesser on Massengale than Ash Log, whereas basal area is greater on Massengale than Ash Log. Continued prescribed fire treatments, including growing season fires, on Massengale Mountain are likely to be effective at maintaining and enhancing the existing Golden-winged habitat and sustaining the Golden-winged breeding population. In some areas, the need to reduce basal area may warrant tree removal. Woody growth may also warrant herbicide use, in which case vegetation composition dictates application procedures (e.g. chemical compound and full-broadcast or spot spraying; Harper 2007). Given the range-wide decline of Golden-wingeds and the potential listing of the species, continued monitoring of the North Cumberland population is warranted.

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CONCLUSION

Reclaimed coal mine lands throughout Appalachia are appealing for conservation of Golden-winged Warblers because the disturbance and subsequent reclamation procedures can create openings of early successional plant communities in an otherwise forested landscape (Canterbury et al. 1996, Bulluck and Buehler 2008, Patton et al. 2010). Furthermore, succession of plant communities on reclaimed coal mines is seriously retarded, relative to forest regeneration following timber harvest. On my 10 mountain-top study sites in the North Cumberland Wildlife Management Area, Tennessee, coal surface-mine reclamation occurred from about 1980–1990 (Bulluck and Buehler 2008). I detected no increase in either Blue-winged Warblers or Blue-winged/Golden-winged hybrids on these sites. Abundance of Golden-wingeds did vary, however, across these sites. The population remained relatively stable on two sites, largely in absence of active habitat management. Decline of the population on three sites was correlated with an increase in sapling height and vine cover, as well as a decrease in shrub/*Rubus* spp. cover.

I documented a population increase from ~8 individuals in 2003 to 24 in 2011 on a single site that was managed annually with prescribed fire, 2007-2011. Furthermore, I detected no negative relationship between daily nest survival and fire history. An increase in Golden-winged abundance following prescribed fire treatments is encouraging, but additional research is needed. Specifically, I recommend replication of prescribed fire on other study sites. Also, future research should evaluate how timing (i.e. season), severity, and return intervals of fire treatments on reclaimed coal mining land affect Golden-winged habitat and nesting success.

APPENDIX

Tables

Table 1.1: Presence-absence surveys conducted in known Golden-Winged Warbler territories, North Cumberland Wildlife Management Area, Tennessee, 2011. Territories were delineated throughout the breeding season, and were based on a minimum of five observations. All surveys were conducted between sunrise and 11:00 am. Overall probability for a given time period was estimated as the mean of probabilities for all surveys combined (n = 4 for round 1 and round 2).

Survey round	Date	Total known territories	Total detected	Probability	SE
1	June 05 - 08	44	29	67.4	5.1
2	June 10 - 15	44	20	50.6	10.4

Table 1.2: Number of individual male Golden-wingeds detected on 10 mountain top study sites in the North Cumberland Wildlife Management Area, Tennessee, 2003-2011. Nest searching was conducted on Ash Log Mtn. from 2003 through 2011; nest searching was conducted on Massengale Mtn. from 2006 through 2011; nest searching was conducted on Bootjack Mtn., Fork Mtn., Burge Mtn. and Red Oak Mtn. 2003 through 2006. Where nest searching was not conducted, abundance counts from presence-absence surveys were corrected by a detection probability factor of 0.59.

Site	**Size (ha)**	**Avg. Elevation (m)**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**	**2011**
Ash Log Mtn.	125	875	16	29	38	36	30	31	37	31	38
Massengale Mtn.	162	830	8.47	5.08	10.17	12	15	13	18	24	24
Bootjack Mtn.	50	818	11	13	9	5	15.25	13.56	8.47	6.78	18.64
Fork Mtn.	40	813	8	12	14	13	10.17	13.56	10.17	10.17	5.08
Burge Mtn.	50	964	7	15	17	13	13.56	13.56	6.78	10.17	6.78
Red Oak Mtn.	120	830	11	13	9	5	6.78	3.39	0	0	0
McNew Gap	135	667	5.08	6.78	13.56	16.95	13.56	10.17	10.17	6.78	5.08
Anderson Mtn.	80	750	.	.	0	0	.	0	0	3.39	3.39
Brushy Mtn.	80	812	.	.	0	3.39	0	0	5.08	6.78	0
Hatfield Knob	73	650	.	.	13.56	8.47	6.78	1.69	6.78	1.69	0
Total			66.56	93.86	124.29	112.81	111.10	99.93	102.46	100.76	100.98

Table 1.3: Mitochondrial DNA haplotype association of Golden-winged Warblers, North Cumberland Wildlife Management Area, Tennessee. Numbers indicate individuals that appeared, phenotypically, to be Golden-winged Warblers. ABW = ancestral Blue-winged Warbler mitochondrial DNA haplotype group; AGW = ancestral Golden-winged Warbler mitochondrial DNA haplotype group. No females were tested during 2003-2005.

	Male		**Female**	
Year	**ABW**	**AGW**	**ABW**	**AGW**
2003-2005	13	80	--	--
2010-2011	4	35	0	12

Table 1.4: Number of nests monitored for each Golden-winged Warbler (GW), Blue-winged Warbler (BW), and hybrid (HY) pair located in the North Cumberland Wildlife Management Area, Tennessee, 2009-2011. All possible combinations; presented as male x female. Hybrid status was determined by phenotype. Males with Golden-winged Warbler phenotypes but Blue-winged Warbler songs were also designated hybrids (n = 2).

Year	GWxGW	GWxHY	HYxGW	HYxHY	GWxBW	BWxGW	BWxHY	HYxBW	BWxBW	Total nests
2009	24	3	0	1	0	1	0	0	0	29
2010	26	5	4	1	0	0	0	1	0	37
2011	20	4	1	0	0	0	0	1	1	27
Total	70	12	5	2	0	1	0	2	1	93

Table 1.5: Estimates of daily nest survival rates, clutch size, and young fledged per successful nest for pairs of phenotypically pure Golden-winged Warblers, pairs including at least one Blue-winged Warbler, and pairs including at least one hybrid in the North Cumberland Wildlife Management Area, Tennessee, 2009-2011. True probability of a nest surviving from initiation to completion is calculated as DSR^n, where n, the total length of the nesting period, equals 25 days for Golden-winged Warblers.

Pairs	**N**	**DSR**	**SE**	**Clutch size**	**Young fledged / successful nest**
Golden-winged	70	0.9607	0.0067	4.46	4.30
Blue-winged	4	0.9393	0.0416	4.25	5.00
Hybrid	19	0.9532	0.0132	4.24	4.14

Table 1.6: Yearly estimates of Golden-winged Warbler daily nest survival rates in the North Cumberland Wildlife Management Area, Tennessee, 2009-2011. True probability of a nest surviving from initiation to completion is calculated as DSR^n. Where n, the total length of the nesting period, equals 25 days for Golden-winged Warblers.

			95% CI	
Year	DSR	SE	Lower	Upper
2009	0.9149	0.0182	0.8716	0.9444
2010	0.9781	0.0060	0.9626	0.9872
2011	0.9521	0.0130	0.9193	0.9720

Table 2.1: Descriptive statistics for 14 habitat variables recorded inside male Golden-winged Warbler territories (n = 77), North Cumberland Wildlife Management Area, Tennessee, 2009-2011. Representative means for each territory were first obtained from thirty 1-m, six 5-m and six 11.3-m plot measurements.

Territory	Paramater	Median	Range
1 m percent ground cover	Forb	44.3	11.8 - 76.0
	Grass	18.4	1.83 - 53.2
	Rubus spp.	11.4	5.43 - 48.9
	Litter	6.5	0.0 - 24.7
	Vine	2.7	0.0 - 30.2
	Course woody debris	0.5	0.0 - 12.6
1 m percent midstory cover	Shrubs <1 m in height	0.0	0.0 - 7.3
	Shrubs >1 m in height	0.0	0.0 - 8.8
	Saplings	16.2	1.3 - 57.2
1 m percent canopy cover	Canopy	8.7	0.0 - 44.7
5 m	No. trees with DBH <10 cm	11.7	1.0 - 59.0
11.3 m	Basal area (m^2/ha)	2.9	0.0 - 18.0
	No. Snags	0.2	0.0 - 7.7
Distance to microedge (m)	Microedge	2.7	0.9 - 6.6

Table 2.2: Means (± SE) of 11 habitat measurements recorded inside male Golden-winged Warbler territories ("used") and means recorded on unused plots on two study sites, Ash Log and Massengale mountains, North Cumberland Wildlife Management Area, Tennessee, 2009-2011.

		Ash Log			Massengale		
Scale	Parameter	Used (n = 51)	Unused (n = 23)	P	Used (n = 26)	Unused (n = 26)	P
1 m Radius	Percent cover forb*	36.13(2.08)	38.53(7.13)	0.639	34.22(3.94)	49.52(2.85)	0.010
	Percent cover grass*	20.32(1.79)	21.75(3.05)	0.552	15.95(1.92)	25.27(2.99)	0.012
	Percent cover *Rubus* spp.*	17.42(2.33)	9.78(2.03)	0.044	10.47(2.01)	8.53(2.41)	0.196
	Percent cover shrubs <1 m in height	1.09(0.71)	0.45(0.22)	0.838	1.68(0.71)	0.59(0.27)	0.189
	Percent cover shrubs >1 m in height†	4.53(1.63)	9.36(4.31)	0.377	7.93(1.33)	1.45(0.50)	< 0.001
	Percent cover sapling*	18.33(2.10)	22.08(5.24)	0.744	17.76(2.64)	9.19(1.68)	0.038
	Percent cover canopy†	9.80(1.51)	12.26(3.51)	0.669	14.35(3.14)	31.20(4.14)	0.002
	Distance to microedge (m)*	2.79(0.23)	2.22(0.52)	0.166	2.62(0.42)	5.62(0.87)	0.003
5 m Radius	Number of trees with DBH <10 cm	17.79(2.01)	22.77(4.25)	0.870	12.61(2.07)	8.74(2.16)	0.052
11.3 m Radius	Basal area (m^2/ha)†*	2.6(0.03)	7.98(0.14)	0.001	6.43(0.09)	12.08(0.16)	0.007
	Number of snags†	0.31(0.16)	1.4(0.42)	<0.001	1.30(0.26)	1.42(0.35)	0.474

* Significant difference at the $P \leq 0.05$ on one site.

† Significant difference at the Dunn-Sidak adjusted $\alpha \leq 0.00465$ on one site.

No habitat variables were significant at the Dunn-Sidak adjusted α across both sites.

Table 2.3: Means (± SE) of 14 habitat measurements recorded inside plots centered on Golden-winged Warbler nests and means recorded inside non-nest plots on two study sites, Ash Log and Massengale Mountains, North Cumberland Wildlife Management Area, Tennessee, 2009-2011. No habitat variables were significant at the Dunn-Sidak adjusted α ≤ 0.0037 for either site.

		Ash Log			**Massengale**		
Scale	Parameter	Nest (n = 68)	Non-nest (n = 70)	P	Nest (n = 24)	Non-nest (n = 24)	P
1 m radius	Percent cover forb*	53.7(3.00)	45.8(3.75)	0.101	64.6(4.01)	48.8(5.54)	0.050
	Percent cover grass	22.8(2.36)	22.9(3.14)	0.532	12.2(2.50)	10.3(2.41)	0.422
	Percent cover woody*	19.9(2.38)	17.3(3.25)	0.019	16.3(3.09)	21.3(5.50)	0.859
11.3 m radius	Percent cover forb	88.9(1.52)	85.5(1.77)	0.097	87.1(2.42)	90.4(2.33)	0.259
	Percent cover grass	67.3(2.72)	69.1(2.71)	0.625	56.9(5.18)	51.7(5.30)	0.598
	Percent cover *Rubus* spp.	50.2(3.25)	45.6(3.47)	0.340	40.8(5.28)	47.3(5.80)	0.515
	Percent cover shrubs*	1.4(0.35)	1.1(0.34)	0.289	13.5(2.75)	10.2(3.88)	0.044
	Percent cover vines	30.2(3.17)	27.2(2.99)	0.473	18.3(3.33)	25.2(4.89)	0.442
	Percent cover saplings >1 m	13.7(1.51)	11.2(1.32)	0.259	10.0(2.64)	12.3(2.67)	0.327
	Percent cover sapling <1 m*	8.2(1.32)	4.6(0.68)	0.026	9.2(2.75)	9.8(2.36)	0.880
	Percent cover canopy*	57.1(3.19)	46.0(3.92)	0.038	42.5(5.38)	49.4(6.46)	0.470
	Basal area (m^2/ha)	4.5(0.59)	3.5(0.71)	0.081	4.7(1.07)	6.93(1.55)	0.376
	Number of snags	0.21(0.06)	0.16(0.08)	0.121	0.83(0.22)	1.46(0.35)	0.268
	Percent density board covered*	81.9(1.73)	72.7(2.33)	0.007	80.0(2.90)	75.8(4.92)	0.844

* Significant difference at the P ≤ 0 .05 on one site.

Table 2.4: Means (± SE) of 10 habitat variables recorded inside plots centered on Golden-winged Warbler nests during the 2005-2006 and re-measurements from a subsample during 2010, North Cumberland Wildlife Management Area, Tennessee. *Rubus* spp. was not recorded separate from shrubs 2005-2006.

		2005 - 2006	2010	
Scale	Vegetation parameter	Nests (n = 27)	Re-measured Nests (n = 10)	P
11.3 m radius	Percent cover forb	80.8(3.6)	79.0(4.1)	0.1820
	Percent cover grass	66.4(3.9)	62.0(6.8)	0.3350
	Percent cover *Rubus* spp.	--	11.5(3.0)	--
	Percent cover shrubs†	43.1(4.53)	3.5(2.5)	<0.0001
	Percent cover vines*	22.7(4.39)	44.5(9.65)	0.0215
	Percent cover sapling >1m†	26.9(3.77)	8.0(2.6)	0.0018
	Percent cover sapling <1 m†	11.4(2.38)	1.0(0.67)	0.0006
	Percent cover canopy	50.5(4.96)	64.0(4.71)	0.0556
	Basal area (m^2/ha)	6.9(1.03)	10.0(2.93)	0.3024
	Average sapling height (m)†	1.8(0.23)	3.2(0.31)	<0.0001

* Significant difference at the $P \leq 0.05$ on one site.

† Significant difference at the Dunn-Sidak adjusted $\alpha \leq 0.0051$ on one site.

Table 2.5: Model notation used in succeeding results tables for models of Golden-winged Warbler daily nest survival rates in the North Cumberland Wildlife Management Area, Tennessee, 2009-2011.

Model	Notation
1) Single estimate of daily survival	$S_{(.)}$
2) Categorical year effect	$S_{(Year)}$
3) Linear time trend effect	$S_{(T)}$
4) Quadratic time trend effect	$S_{(T^2)}$
5) Nest age effect	$S_{(A)}$
6) Quadratic effect of nest age	$S_{(A^2)}$
7) Cubic effect of nest age	$S_{(A^3)}$
8) Maximum daily temperature effect	$S_{(MaxTemp)}$
9) Minimum daily temperature effect	$S_{(MinTemp)}$
10) Daily precipitation effect	$S_{(Precip)}$

Table 2.6: Model selection results for the logistic-exposure models of daily nest survival rates for Golden-winged Warblers in the North Cumberland Wildlife Management Area, Tennessee, 2009-2011. K is the number of parameters in the model, AICc is Akaike's Information Criterion for small samples, ΔAICc is the scaled value of AICc, w_i is the Akaike weight.

Model	K	AIC_c	ΔAIC_c	w_i	Model Likelihood
$S_{(\text{Year} + T^2)}$	5	297.52	0.00	0.454	1.000
$S_{(\text{Year} + T^2 + \text{MaxTemp})}$	6	299.52	2.01	0.166	0.367
$S_{(\text{Year} + T^2 + \text{Age}^2)}$	7	299.98	2.47	0.132	0.291
$S_{(\text{Year} + \text{MaxTemp})}$	4	300.76	3.25	0.090	0.197
$S_{(\text{Year})}$	3	301.84	4.33	0.052	0.115
$S_{(\text{Year} + T^2 + \text{Age}^2 + \text{MaxTemp})}$	8	302.00	4.48	0.048	0.106
$S_{(\text{Year} + \text{Age}^2)}$	5	302.32	4.81	0.041	0.091
$S_{(T^2)}$	3	305.75	8.24	0.007	0.016
$S_{(T^2 + A^2)}$	5	306.93	9.41	0.004	0.009
$S_{(T)}$	2	308.56	11.05	0.002	0.004
$S_{(T^2 \text{ x } A^2)}$	6	308.94	11.43	0.002	0.003
$S_{(T \text{ x } A^2)}$	5	310.61	13.10	0.001	0.001
$S_{(\text{MaxTemp})}$	2	313.49	15.98	0.000	0.000
$S_{(\text{Age}^2)}$	3	313.56	16.04	0.000	0.000
$S_{(.)}$	1	313.64	16.13	0.000	0.000
$S_{(\text{Age})}$	2	313.68	16.17	0.000	0.000
$S_{(\text{MinTemp})}$	2	314.97	17.45	0.000	0.000
$S_{(\text{Age}^3)}$	4	315.51	18.00	0.000	0.000
$S_{(\text{Precip})}$	2	315.63	18.12	0.000	0.000

Table 2.7: Model selection results for the effect of site and prescribed burning on daily nest survival rates for Golden-winged Warblers in the North Cumberland Wildlife Management Area, Tennessee, 2009-2011. Nests were assigned to one of three burn history groups, 0-1 year post burn (n = 12), ≥2 years post burn (n = 12), control (no burn; n = 68). K is the number of parameters in the model, AICc is Akaike's Information Criterion for small samples, ΔAICc is the scaled value of AICc , w_i is the Akaike weight. S(.) is the constant survival model.

Model	K	AIC_c	ΔAIC_c	w_i	Model Likelihood
$S_{(.)}$	1	313.64	0.00	0.664	1.000
$S_{(\text{Site})}$	2	315.65	2.00	0.244	0.367
$S_{(\text{Burn history})}$	3	317.59	3.95	0.092	0.139

Table 2.8: Parameter estimates for the effect of site and prescribed burning on daily nest survival rates (DSR) for Golden-winged Warblers in the North Cumberland Wildlife Management Area, Tennessee, 2009-2011.

		DSR estimate	SE	95% CI Lower	95% CI Upper
Site	Ash Log (n = 68)	0.9584	0.0070	0.9423	0.9701
	Massengale (n = 24)	0.9576	0.0120	0.9268	0.9758
Prescribed burns	No burn (n = 68)	0.9591	0.0069	0.9433	0.9706
	0-1 year post burn (n = 12)	0.9552	0.0179	0.9039	0.9797
	≥2 years post burn (n = 12)	0.9553	0.0179	0.9041	0.9798

Table 2.9: Model selection results for the effect of habitat variables on daily nest survival rates for Golden-winged Warblers in the North Cumberland Wildlife Management Area, Tennessee, 2009-2011. K is the number of parameters in the model, AICc is Akaike's Information Criterion for small samples, ΔAICc is the scaled value of AICc, w_i is the Akaike weight. S(.) is the constant survival model.

Model	K	AIC_c	ΔAIC_c	w_i	Model Likelihood
S(*Rubus* spp. in nest substrate)	2	310.30	0.00	0.237	1.000
S(% *Rubus* spp. cover within 1 m)	2	312.48	2.19	0.091	0.386
S(Distance to forest edge)	2	313.12	2.82	0.079	0.335
S(*Rubus* spp. cover within 11.3 m)	2	313.54	3.24	0.058	0.244
S(.)	1	313.64	3.35	0.047	0.198
S(*Solidago* spp. cover within 11.3 m)	2	313.82	3.53	0.049	0.172
S(*Solidgao* spp. in nest substrate)	2	314.46	4.16	0.036	0.125
S(Vine cover within 11.3 m)	2	314.50	4.21	0.035	0.122
S(Forb cover within 1 m)	2	314.76	4.47	0.030	0.107
S(Forb cover within 11.3 m)	2	314.93	4.63	0.028	0.099
S(Shrub cover within 11.3 m)	2	315.07	4.78	0.026	0.092
S(Tree sapling in nest substrate)	2	315.12	4.83	0.025	0.090
S(Grass cover within 1 m)	2	315.18	4.89	0.025	0.087
S(Grass cover within 11.3 m)	2	315.27	4.98	0.024	0.083
S(*Sirecea lespedaza* in nest substrate)	2	315.36	5.06	0.023	0.080

Table 2.9 (cont.)

$S_{\text{(Canopy cover within 11.3 m)}}$	2	315.39	5.09	0.022	0.078
$S_{\text{(Density board cover)}}$	2	315.57	5.27	0.020	0.072
$S_{\text{(Grasses in nest substrate)}}$	2	315.61	5.31	0.020	0.070
$S_{\text{(Vine cover within 1 m)}}$	2	315.63	5.33	0.020	0.070
$S_{\text{(Saplings B within 11.3 m)}}$	2	315.64	5.35	0.020	0.069
$S_{\text{(Saplings A within 11.3 m)}}$	2	315.65	5.35	0.020	0.069
$S_{(\textit{Sirecea lespedeza}\text{ within 11.3 m})}$	2	315.65	5.35	0.020	0.069

Table 2.10: Beta estimates and 95% confidence intervals (CI) for habitat parameters in the top models for nest survival of Golden-winged Warblers in the North Cumberland Wildlife Management Area, Tennessee, 2009-2011.

			95% CI	
Parameter	β Estimate	SE	Lower	Upper
Rubus spp. in nest substrate	-0.7887	0.3249	-1.4255	-0.1518
% *Rubus* spp. cover (1-m)	-0.0147	0.0078	-0.0300	0.0006
Distance to forest edge	-0.0077	0.0045	-0.0164	0.0010
Rubus spp. cover (11.3-m)	-0.0083	0.0057	-0.0195	0.0029

Table 2.11: Model selection results for the effect of temporal, site and habitat variables on daily nest survival rates for Golden-winged Warblers in the North Cumberland Wildlife Management Area, Tennessee, 2009-2011. K is the number of parameters in the model, AICc is Akaike's Information Criterion for small samples, ΔAICc is the scaled value of AICc, w_i is the Akaike weight. S(.) is the constant survival model.

Model	K	AIC_c	ΔAIC_c	w_i	Model Likelihood
S(Year + T^2 + *Rubus* spp. in nest substrate)	6	295.22	0.00	0.271	1.000
S(Year + T^2 + *Rubus* spp. within 1 m)	6	295.23	0.01	0.269	0.993
S(Year + T^2 + *Rubus* spp. in nest substrate + Distance to forest edge)	7	296.92	1.70	0.115	0.427
S(Year + T^2 + *Rubus* spp. in nest substrate + Site)	7	297.24	2.02	0.098	0.364
S(Year + T^2)	5	297.52	2.30	0.086	0.317
S(Year + T^2 + *Rubus* spp. within 11.3 m)	6	298.79	3.57	0.045	0.168
S(Year + T^2 + *Rubus* spp. in nest substrate + Distance to forest edge + Site)	8	298.95	3.73	0.042	0.155
S(Year + T^2 + Distance to forest edge)	6	299.45	4.24	0.033	0.120
S(Year + T^2+ Maximum temperature)	6	299.52	4.31	0.031	0.116
S(Year)	3	301.84	6.62	0.010	0.036
S(*Rubus* spp. in nest substrate)	2	310.30	15.08	0.000	0.001
S(*Rubus* spp. within 1 m)	2	312.48	17.26	0.000	0.000
S(Distance to forest edge)	2	313.12	17.90	0.000	0.000
S(*Rubus* spp. within 11.3 m)	2	313.54	18.32	0.000	0.000
S(.)	1	313.64	18.43	0.000	0.000
S(Site)	2	315.65	20.43	0.000	0.000

Figures

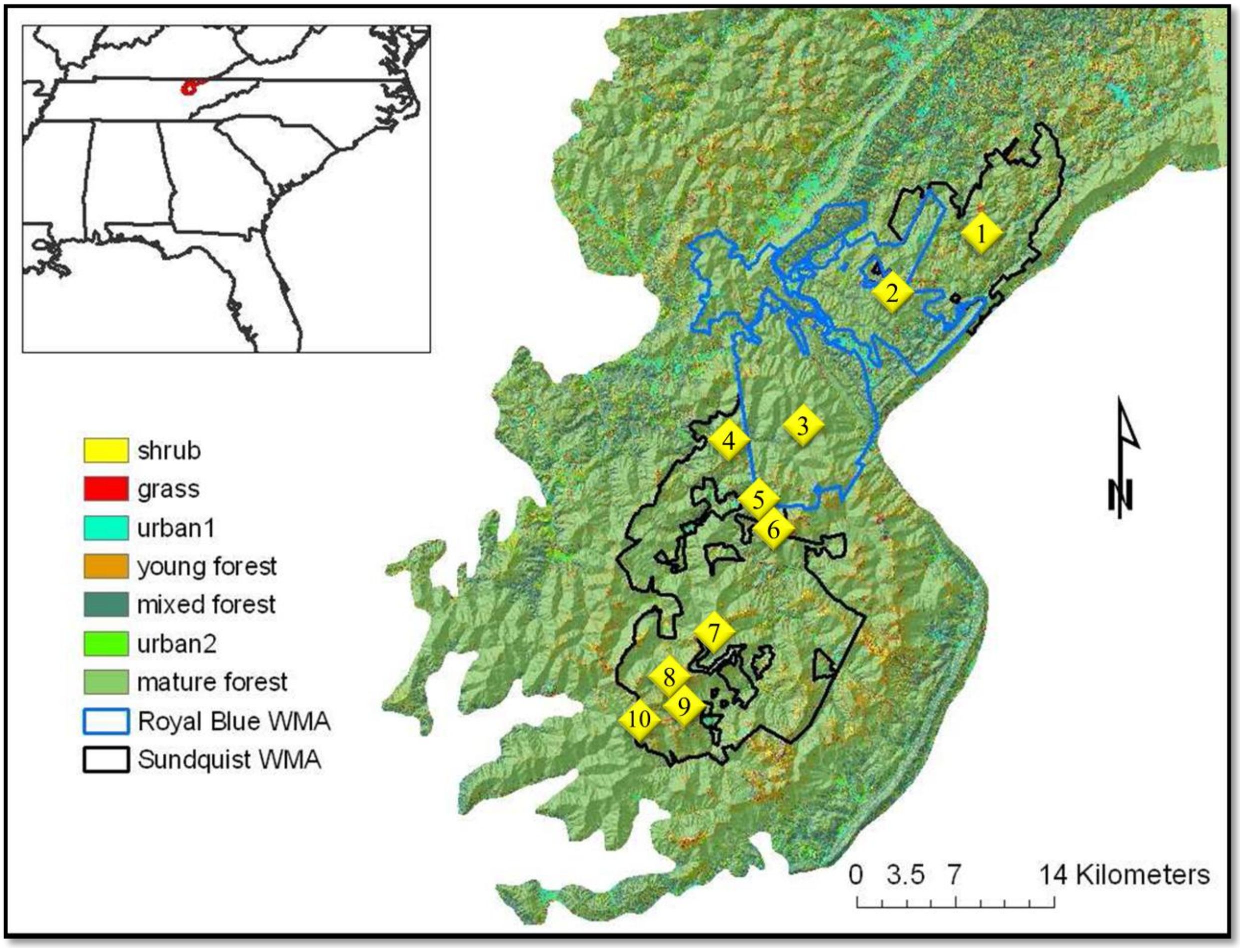


Figure 1.1: Landcover map of the Cumberland Mountains in northeastern Tennessee (from classified Landsat TM satellite image, September 2000). Royal Blue Wildlife Management Area (WMA) and Sundquist WMA have combined to become the North Cumberland WMA. The 10 study sites were: 1) Hatfield Knob 2) McNew Gap. 3) Brushy Mtn. 4) Anderson Mtn. 5) Massengale Mtn. 6) Ash Log Mtn. 7) Bootjack Mtn. and 8) Fork Mtn. 9) Burge Mtn. and 10) Red Oak Mtn. Nest searching was conducted on only Ash Log Mtn. and Massengale Mtn. Presence-absence surveys were conducted on the 8 additional study sites.

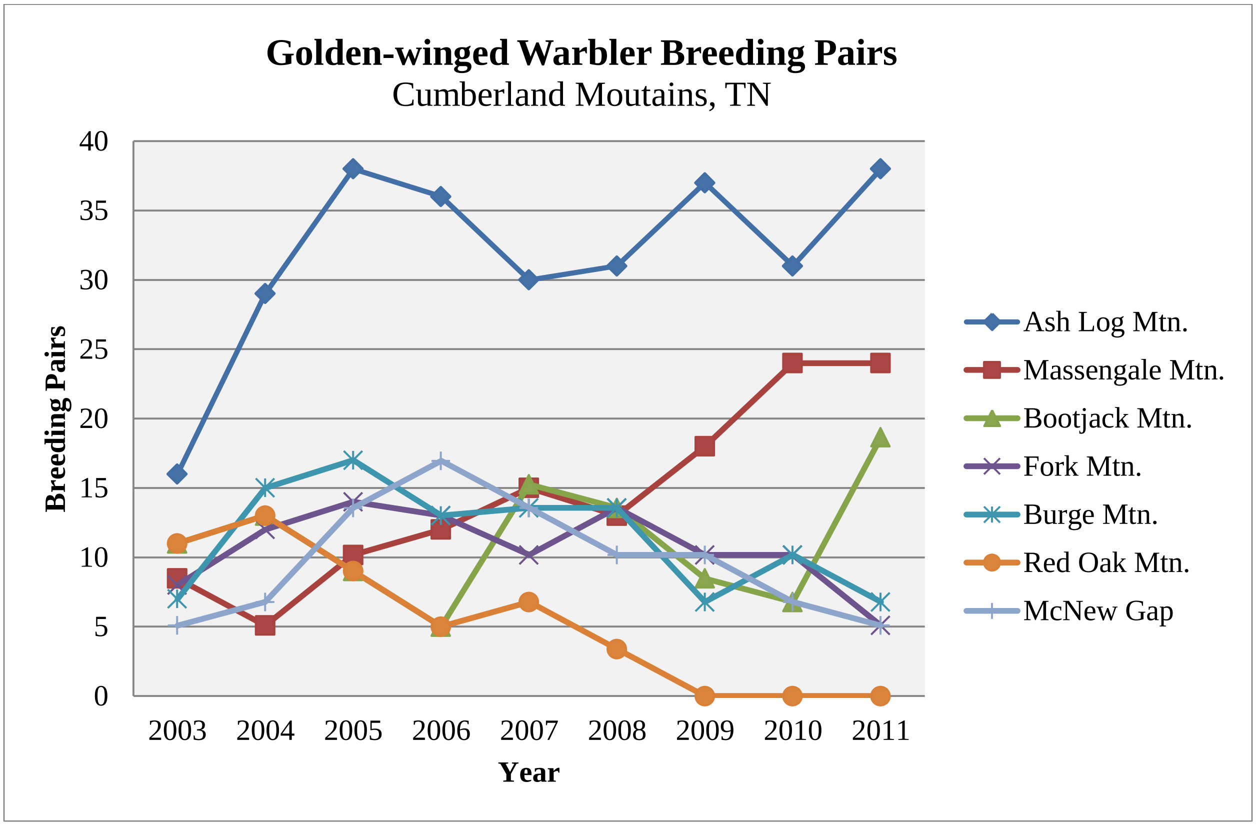


Figure 1.2: Golden-winged Warbler population trends from presence-absence surveys on seven mountaintop sites in the North Cumberland Wildlife Management Area, Tennessee, 2003-2011. Prescribed fire was conducted in a single burn unit on Ash Log during 2007. Prescribed fire was conducted among three burn units on Massengale annually 2007-2011.

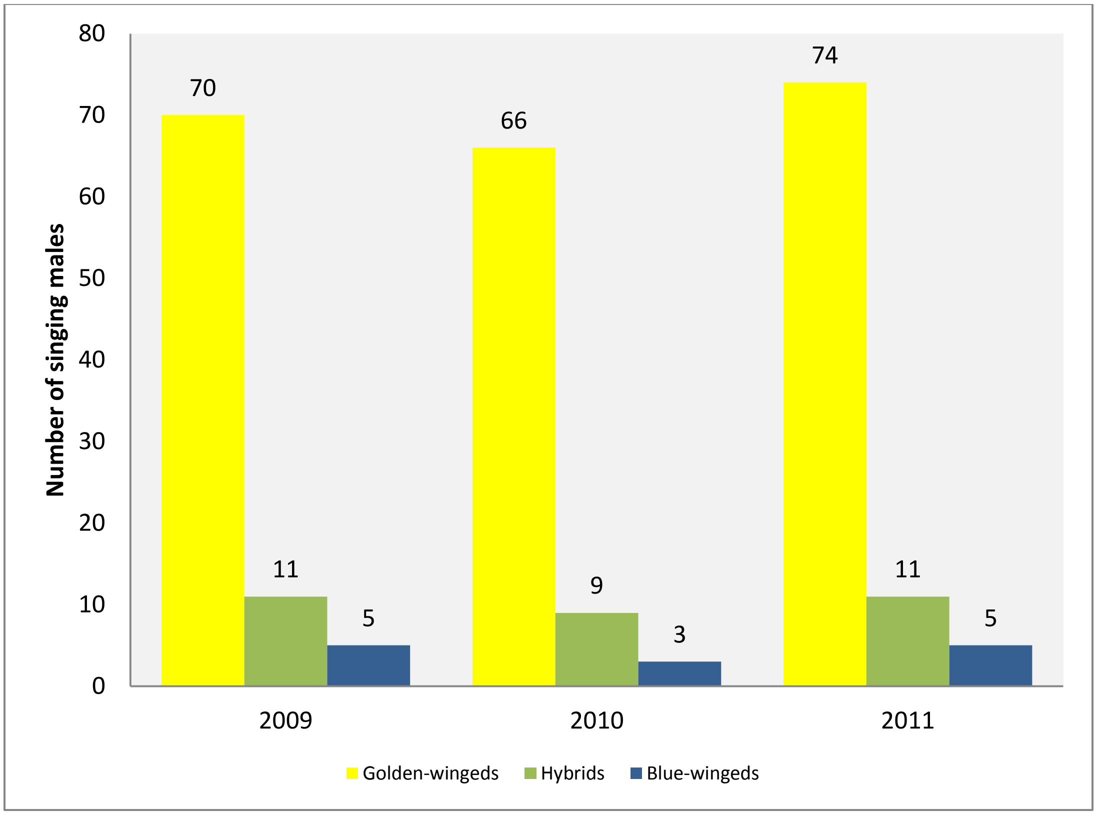


Figure 1.3: Number of singing males detected during presence-absence surveys, North Cumberland Wildlife Management Area, Tennessee, 2009-2011. Individual designations based on phenotype (i.e. plumage characteristics) and/or song.

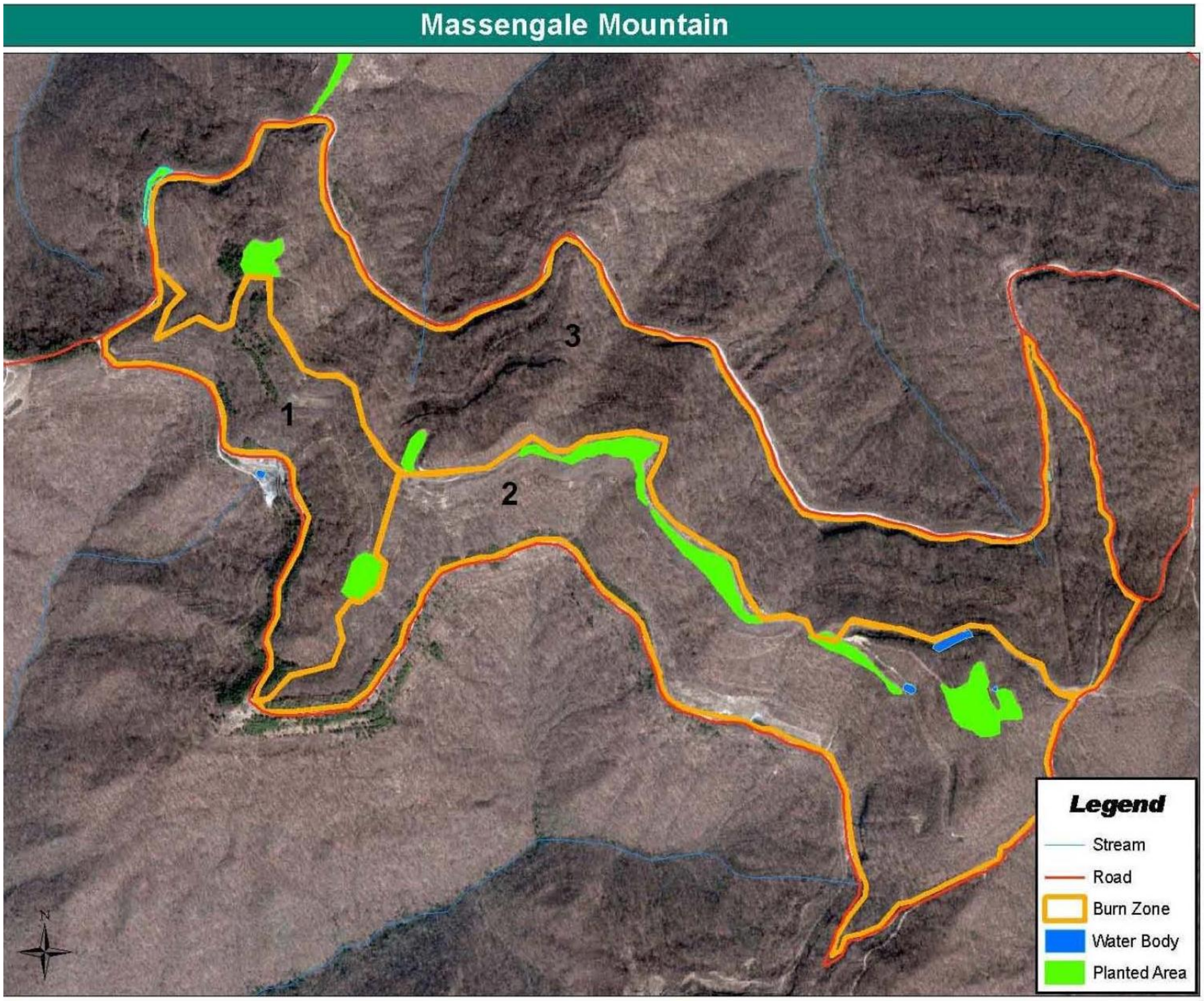


Figure 2.1: Three separate burn zones (outlined in orange) on Massengale Mountain, North Cumberland Wildlife Management Area, Tennessee. Zone 1 (40 ha) was burned in 2008, 2009, and 2011; zone 2 (115 ha) was burned in 2007 and again in 2010; and zone 3 (145 ha) was burned in 2008 and 2011. All burns were conducted in the late winter or early spring prior to leaf-out.

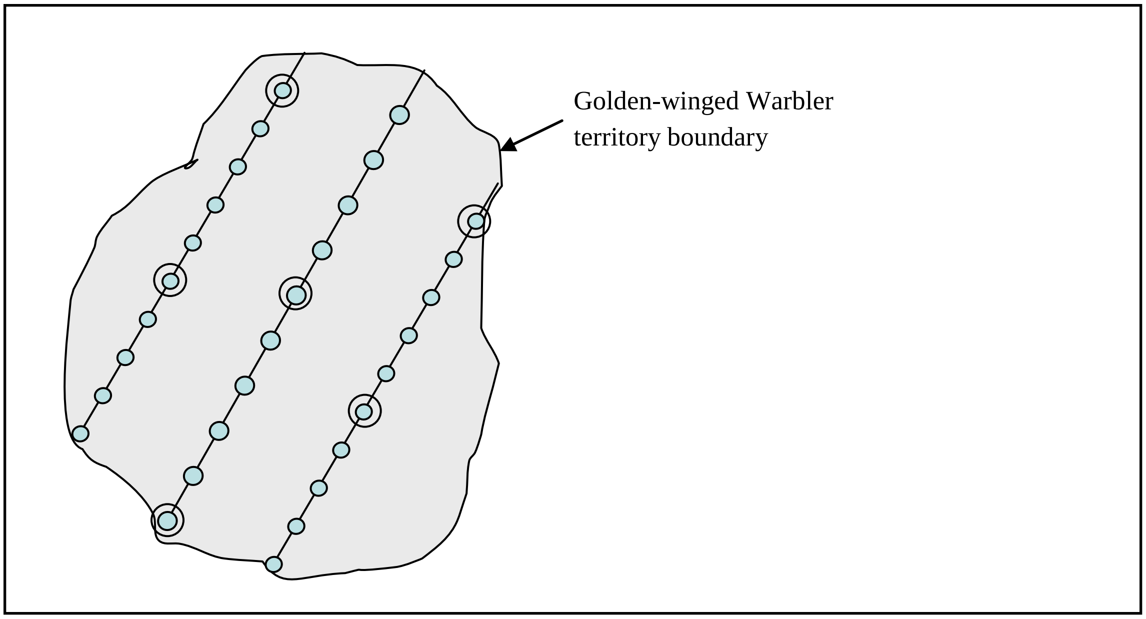


Figure 2.2: Diagram of the vegetation sampling design used to characterize habitat features within Golden-winged Warbler territories. Thirty 1-m vegetation sampling locations (small circles), and 5-m and 11.3-m radius sampling locations (larger circles) were located along line transects.

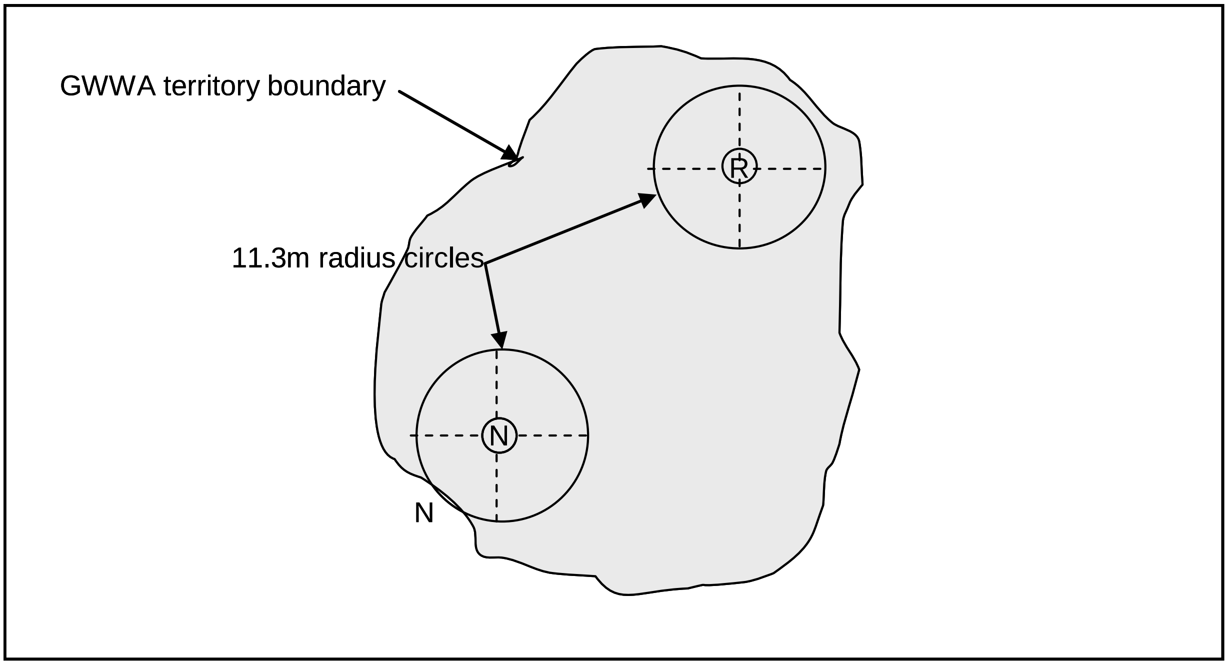


Figure 2.3: Diagram of the vegetation sampling design used to characterize habitat features at each known Golden-winged Warbler nest location (N) and a paired random location (R) within territories. Dashed lines represent 11.3 m transects in each cardinal direction.

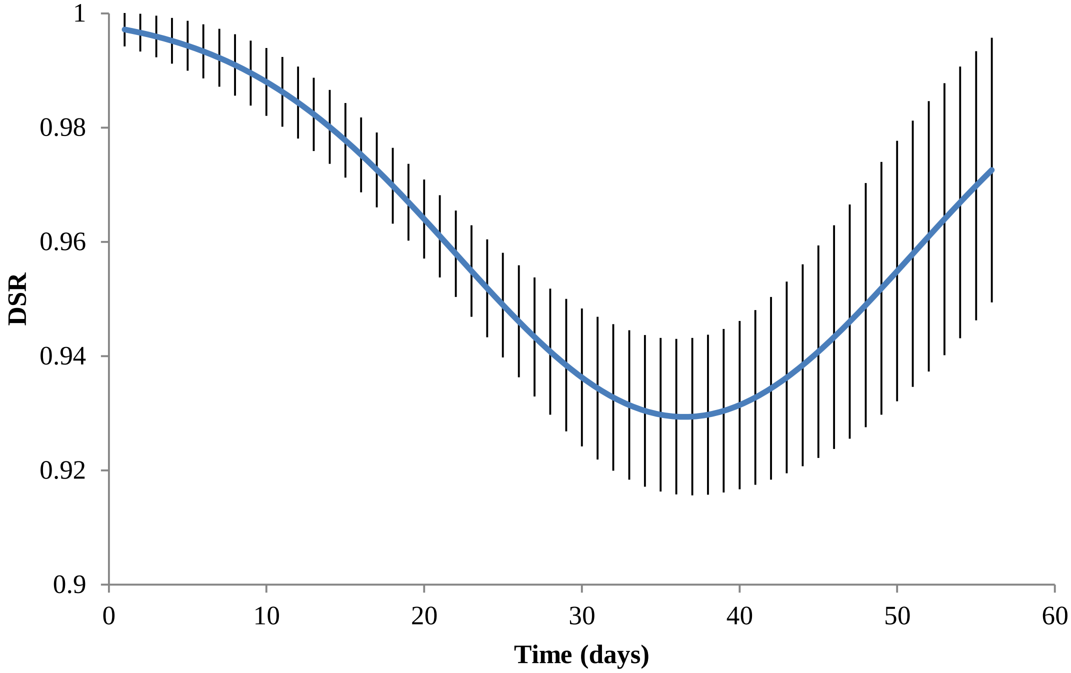
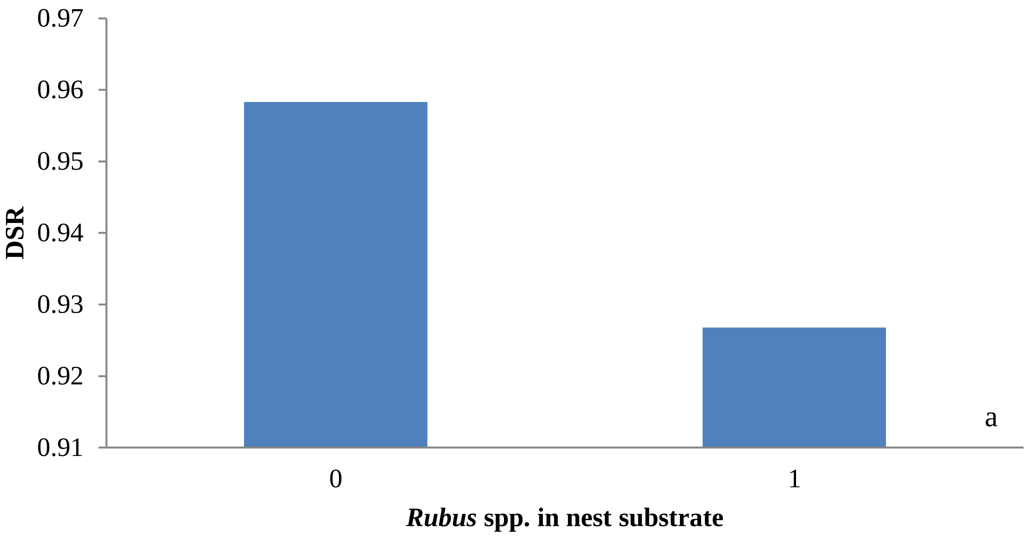
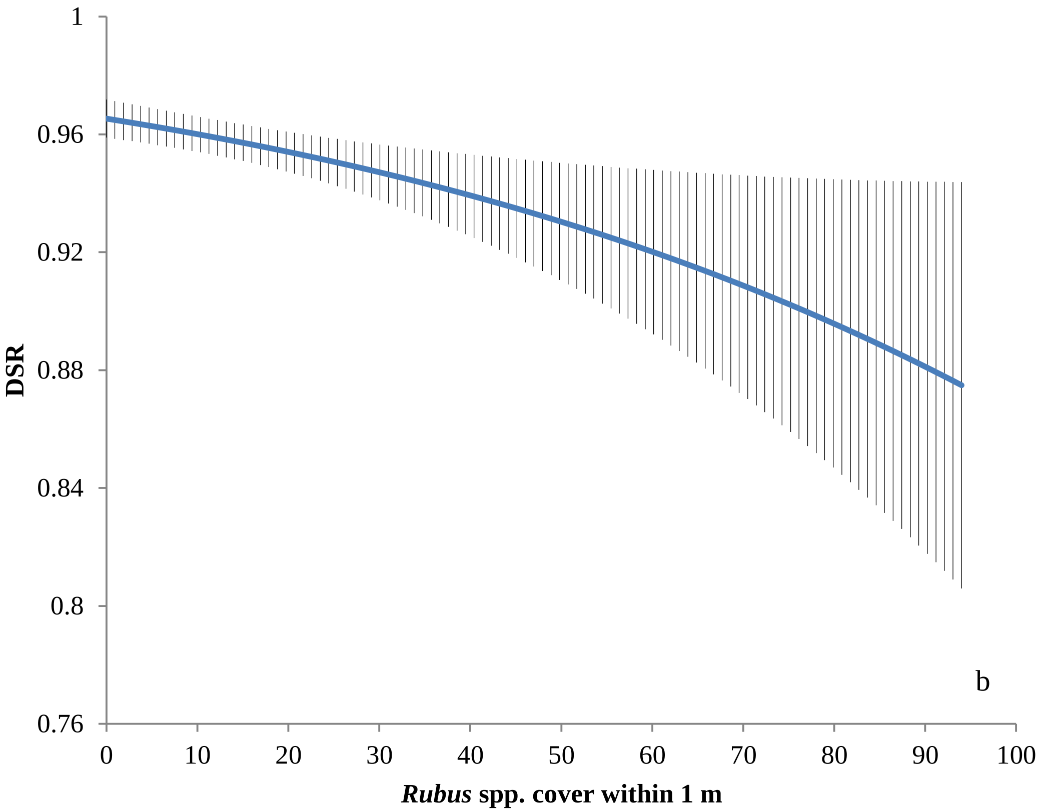


Figure 2.4: Daily survival rate of Golden-winged Warbler nests in relation to day of the nesting season in the North Cumberland Wildlife Management Area, Tennessee, 2009-2011. May 2nd was standardized as day 1. Vertical lines represent standard errors for the logistic-exposure model.


0.97
0.96
0.95
0.94
0.93
0.92
0.91
DSR
0
1
Rubus spp. in nest substrate
a


1
0.96
0.92
0.88
0.84
0.8
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DSR
0
10
20
30
40
50
60
70
80
90
100
Rubus spp. cover within 1 m
b

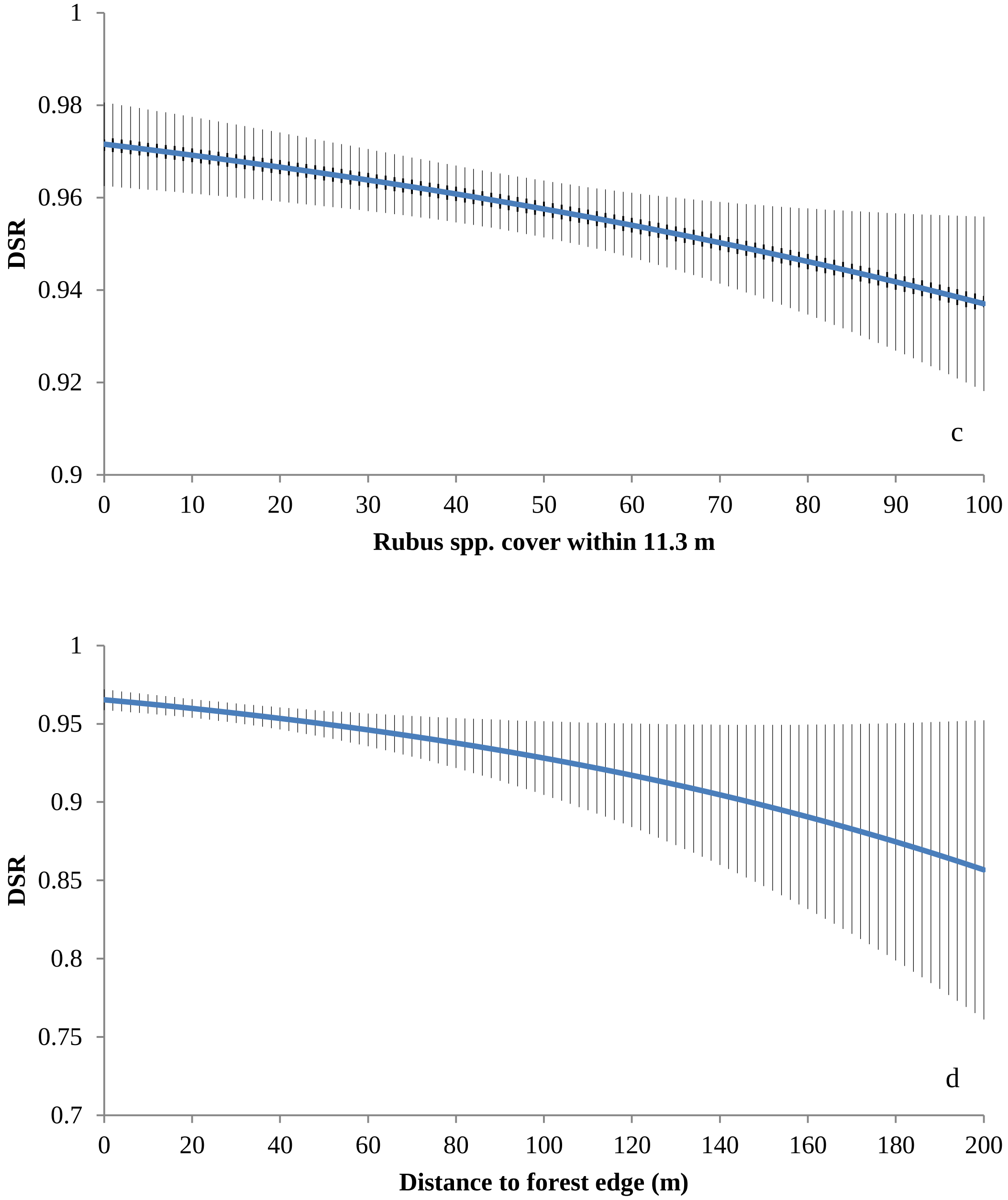

Figure 2.5: Golden-winged Warbler daily nest survival rate (DSR) as a function of nests with (1) and without (0) *Rubus* spp. in the nesting substrate (a), percent *Rubus* spp. cover within 1 m of the nest (b), percent *Rubus* spp. cover within 11.3 m of the nest (c), and distance of nest to a mature forest edge (d), North Cumberland Wildlife Management Area, 2009-2011. Vertical lines represent standard errors.

VITA

Katie Percy, daughter of Adrian and Deborah Percy, was born in Baton Rouge, Louisiana, on May 30, 1984. She graduated from Baton Rouge Magnet High School in 2003, and earned her Bachelor of Science degree in Natural Resource Ecology and Management, with a concentration in Conservation Biology, from Louisiana State University in December 2007. In August 2009 she entered graduate school at the University of Tennessee seeking a master's degree in Wildlife and Fisheries Science. Under the advisement of her major professor, Dr. David A. Buehler, Katie studied the population of Golden-winged Warblers located in the North Cumberland Wildlife Management Area for her thesis research detailed in this document. Through the pursuit of her bachelor's and master's degrees she has gained invaluable experiencing studying populations of wild birds located in a diverse array of ecosystems, including longleaf pine savanna in Mississippi, barrier island chains off the Louisiana coastline, and the Cumberland Mountains in Tennessee.