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AVIAN COMMUNITY RESPONSE TO FIRE-MEDIATED REGENERATION OF NATIVE PINE STANDS IN THE MOUNTAINS OF SOUTH CAROLINA

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AVIAN COMMUNITY RESPONSE TO FIRE-MEDIATED
REGENERATION OF NATIVE PINE STANDS IN THE MOUNTAINS
OF SOUTH CAROLINA

A Thesis
Presented to
The Graduate School of
Clemson University

In Partial Fulfillment
of the Requirements for the Degree
Master of Science
Wildlife and Fisheries Biology

by
Curtis D. Walker
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Accepted by:
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Dr. Julia L. Sharp
Dr. Bo Song
Dr. Geoff Wang

ABSTRACT

To assess the impacts of fire disturbance management on the avian community at Jocassee gorges in the mountains of South Carolina, a total of 1000 10-minute, 50 m radius point counts were conducted in treatment and control plots during the spring breeding seasons of 2011 and 2012. Comparisons of avian communities were made between the burned treatment sites and reference control sites to examine community and priority species response to prescribed fire. Values of species diversity, species richness, and total number of individuals were found to be significantly higher in the burned treatment plots than in the control plots as a result of differences in structural complexity and the distribution of resources. The occurrence of focal species, as well as other species, was found to vary between sites. Species associated with early successional and more open habitats, such as eastern wood-pewees (*Contopus virens*) and indigo buntings (*Passerina cyanea*) were observed more often in burned sites, while species that require shrubbery and broad-leaved foliage on which to forage, like black-throated green (*Dendroica virens*) and hooded warblers (*Wilsonia citrine*), were observed more often in control sites. Models created using structural vegetation data identified characteristics of vegetation and landform that were found to be useful in predicting the occurrence of 6 of the 7 priority species at Jocassee. Differences in the occurrence of nesting and foraging guilds were related to differences in complexity of habitat structure and composition. This research suggests that fire management can be a useful tool to create wider variation across the landscape, providing increased opportunities for nesting and foraging resources for an array of bird species.

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CHAPTER I

INTRODUCTION

In recent decades dating back to the middle of the 20th century, neo-tropical migrant bird populations have exhibited steady declines across North America. Included among these are species that breed within the spectrum of habitats ranging from hydric low lying wetlands to xeric ridge tops of mountainous landscapes (Askins et al. 1987, Robbins et al. 1989b, Terborgh 1989, Askins et al. 1990, Finch 1991, Sauer and Droege 1992). Across the entirety of North America, suitable habitats for migratory birds are increasingly imperiled through ecosystem alteration. Fragmentation and loss of suitable habitat for wintering, breeding, and migration are likely explanations for negative species trends (Robbins et al. 1989). The alarming reality of these trends and the looming potential for extirpation and extinction of specialized species has struck a chord with conservationists and researchers alike. As a result, governmental agencies have taken initiative and partnerships such as Partners in Flight (PIF) and the North American Bird Conservation Initiative (NABCI) have been formed to address the issues. The resulting tools of such partnerships used in the identification of Important Bird Areas (IBAs) and Bird Conservation Regions (BCRs) are integral in species priority diagnoses for the physiographical regions of North America.

In the eastern United States and specifically in the southern Appalachians, early-succession songbirds show the strongest declines of any group of birds (Hunter et al. 2001, Sauer et al. 2005). Partners in Flight has identified a portion of this region, the Blue Ridge Physiographic Region, as an area of importance for breeding bird populations (Hunter et al. 1999). Many global and state IBAs have been designated in the region, including several National Forests (Sumter National Forest) and State Parks (Caesar's Head) to assist in

management and conservation of priority species. For these efforts to be successful, researchers and land managers must begin to uncover the relationships between these birds and their specific habitat requirements (Martin 1992, Robbins et. Al 1989b). For many species, our current knowledge of habitat relation and selection is not comprehensive. Limitations in our understanding are often due to geographical/topographical variation, differences in vegetational composition, climatic stability or instability, and other microsite conditions that have the potential to be highly variable within the breeding range of a given species. For this reason, it is important to develop habitat associations on more regional scales.

It can be difficult to understand true avian-habitat relationships without the implementation of management that best promotes natural and historical operations specific to the region of interest. Ecological communities of the present day rarely exist in their pristine state, and thus are not capable of maximum ecological production, nor is it possible to accurately derive what maximum ecological production might be from such a community without intense research and statistical inference. This requires researchers to identify habitat characteristics of different scales that are related to bird occurrence and to develop models for management that encourage the proliferation of historically accurate communities. In South Carolina, the majority of conservation attention has been focused on coastal plain pine habitats. As Longleaf pine (*Pinus palustris*) forests have received attention from natural resource managers and conservationists, other native pine habitats have gone largely ignored. Native Southern yellow pine habitats also occur throughout the Southern Appalachians, but are local in the mountains (Hunter et al. 1999). Dickson (2001) cites that montane yellow pine habitat, especially Table Mountain Pine (*Pinus pungens*), is vulnerable to further decline because of fire suppression over the past half-century. As this habitat declines, associated avian communities within the slim

montane region of South Carolina may continue to lose diversity. One possible solution to this problem is to integrate avian community information with silvicultural methodologies that would enable land managers to develop relationships between individual species and historical processes specific to the site.

Fire is capable of dramatically altering southern Appalachian forests, providing early-succession and woodland habitats that support declining birds. Land management agencies and the public often view severe fires and canopy mortality negatively; however, these fires provide forest structure not found elsewhere on the landscape (Rush et al. 2011). Many species that require mature forest to nest also use early-succession forests as fledglings (Anders et al. 1998, Vega Rivera et al. 1999, Marshall et al. 2003). Today, early-succession forests are usually provided by human created disturbances such as clear-cuts, power-line rights of way, or roadsides (Hunter et al. 1999). Natural mechanisms that formerly provided this habitat appear to function at reduced levels or are absent from Appalachian landscapes. As a result, many early-succession bird species have become extirpated from the southern Appalachians and others have declined in response to decreased intensity of forest management (Klaus et al. 2005).

In recent years, advancements in computer technologies have enabled researchers and land managers alike to integrate habitat characteristics and the projected consequences of silvicultural practices into virtual environments. Computer visualization software has been used to effectively convey immediate and future impacts of land management to the general public, landowners, and the scientific community. This software combines Digital Elevation Models with forest stand data to develop an accurate picture of current and forecasted habitat conditions following management. The combination of visualization techniques and silvicultural practices

that promote natural site conditions may prove to be useful in the effort to restore historical avian communities to the South Carolina Mountains while maintaining public support.

The objectives of this study were:

1. To compare bird community composition (richness, diversity, evenness, abundance) between ridge top sites treated to encourage the regeneration of native pine species and forests of moderately disturbed control sites.
2. To examine relationships between the presence or absence of priority species (Eastern wood peewee, Ovenbird, Worm-eating warbler, Scarlet tanager, Hooded warbler, Black and white warbler, and Black throated green warbler) and habitat characteristics.
3. To examine associations between individual foraging/nesting guilds and site selection.
4. To provide baseline information on the presence, distribution, and importance of ridge top habitats for mature forest bird species to aid in management and future research.
5. To use computer visualization techniques to effectively convey immediate and future impacts of land management practices to the general public, landowners, and the scientific community.

The null hypotheses considered in this study were:

- H_0 : Avian community composition (richness, diversity, evenness, abundance) does not differ between treatment and control sites.
- H_0 : Occurrence of priority species does not differ between treatment and control sites or among habitat characteristics.
- H_0 : Site selection of foraging guilds does not differ between treatment and control sites.
- H_0 : Site selection of nesting guilds does not differ between treatment and control Sites.

CHAPTER II

METHODS

Study Area

The Jocassee Gorges occur in northern Oconee and Pickens counties in northwestern South Carolina along the South Carolina – North Carolina border (Fig. 1). Lying in the extreme southern section of the Blue Ridge Mountain physiographic province of the southern Appalachians, The Jocassee Gorges occupy the southern chains of mountains that rise abruptly from the lower-elevation Piedmont region of South Carolina (Bowman 1911; Griffith et al. 2002). The topography of the Jocassee Gorges consists of ridges, stream-dissected hill slopes composed of side slopes and convex nose slopes between stream ravines, deep stream gorges, bottoms or steep slopes adjacent to large streams, and flat-floored coves embedded on upper slopes or near the origins of first-order streams. Slope gradients on hill slopes range from 20% to 70% and vary along the hill slope plan. Typical elevations in the study area range from 350 to 850 m, and elevations of the larger streams are 180–250 m lower than upper portions of surrounding hill slopes. Elevations of the Jocassee Gorges are higher than those of the Piedmont region to the south but lower than the elevations of many mountain chains to the north in western North Carolina (Whittaker 1956; Carter et al. 2000). Within the study area, elevation does not impact ecosystem composition (Mowbray and Oosting 1968; Racine 1971) as it does in the higher-elevation mountains of the southern Appalachians (Callaway et al. 1987; McNab et al. 1999; Carter et al. 2000).

Temperatures in the region are moderate (annual average minimum-maximum: (7.6°C-21.7°C), lacking the extremes of heat and drought, and rainfall in the area is high (Cooper and Hardin 1970). Annual precipitation ranges from 160 to 339 cm, averaging 173.72 cm, with the highest periods of rainfall coming in June to August and January to February (USACOE 2010). The region is characterized by a high diversity of flora and fauna as a result of its location at the interface between the Piedmont and Blue Ridge Mountains (Rankin 1998, Abella 2002). Plant communities from both regions are represented along the escarpment, creating a transitional zone from the lower elevation, rolling hills of the Piedmont to higher elevation, rugged peaks of the mountains (Braun 1950). The biotic communities of the Jocassee Gorges have been subjected to a variety of forest management activities beginning in the late 1800s and early 1900s. Fire suppression and timber harvest have taken place on the property since the early to mid-20th century. An estimated 72% of the property has been clear-cut or selectively harvested since 1964, but some remnant mature stands remain (Rankin 1998). In 1998, the state of South Carolina purchased the property from Crescent Resources, a subsidiary of Duke Power, and is currently under the management of the South Carolina Department of Natural Resources (SCDNR).

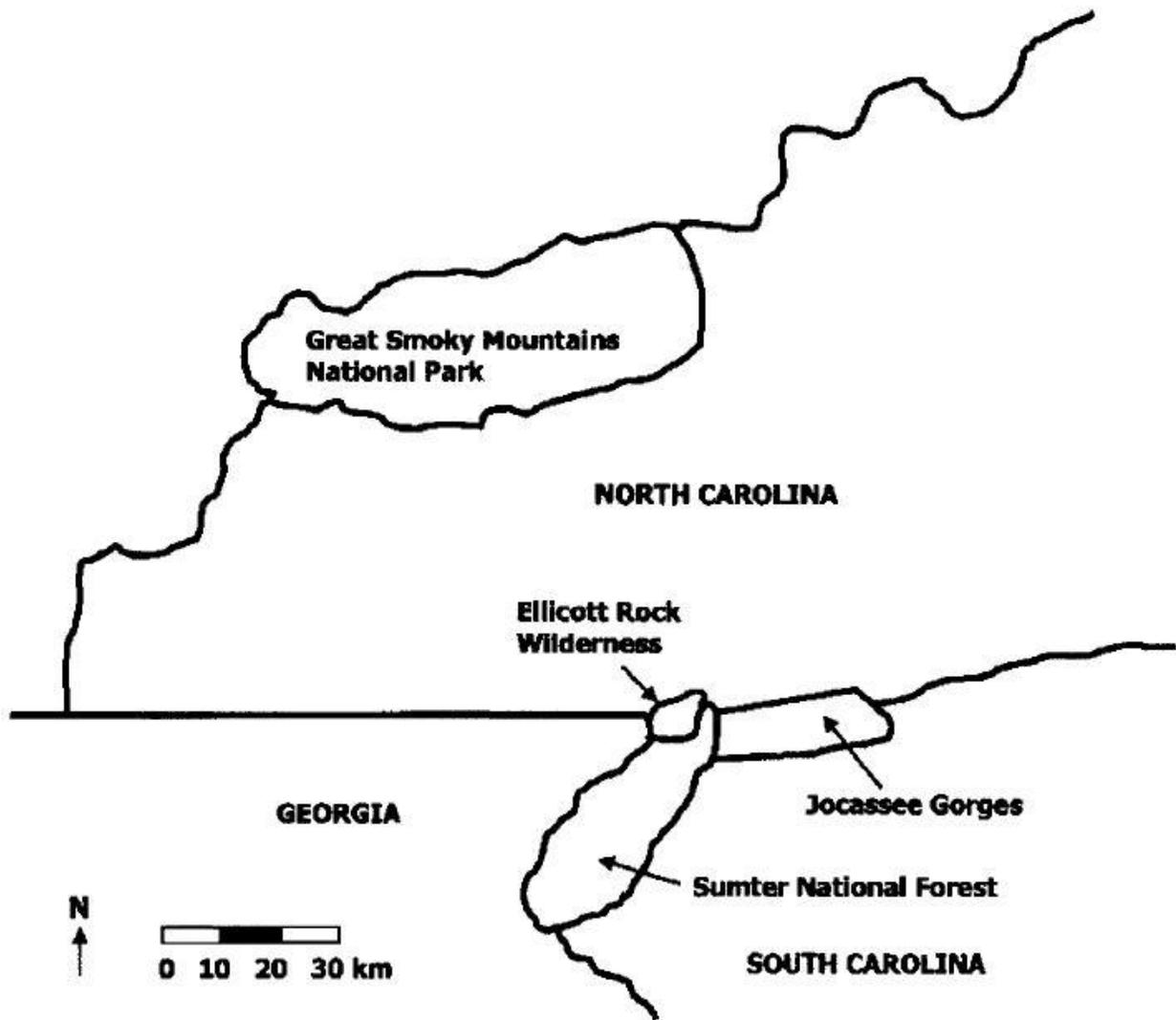


Figure 1. Location of the 13,000 ha Jocassee Gorges, South Carolina. Adapted from Abella (2004)

Point Selection for Avian Sampling

Potential point locations used in this study were identified using digital orthophoto quarter quadrangles and topographic maps delineating stands treated with prescribed fire (treatment) and stands having no history of fire within the last 30 years (control) provided by the South Carolina Department of Natural Resources (SCDNR) and through ground reconnaissance. The treatment area was comprised of two major zones which were most recently burned one year apart in 2010 and 2011, respectively. A total of 100 points were established in both the treatment and control areas on xeric ridge tops of Southeastern to Western middle to upper and convex nose slope positions suitable for the growth and regeneration of native pine stands. The burned areas were separated into 2 individual units including those that were burned in 2006 and 2010 and those that were burned in 2008 and 2011. Points were placed a minimum of 200 m apart to reduce the chance of repeated observations of the same individual or individuals previously sampled. Each point was permanently marked with a handheld GPS (Garmin 60 GSX) for navigational purposes and location accuracy. Criteria for exclusion included poor GPS reliability, points subject to edge-effects between treatment and control areas or those bordering other habitats including streams, power-line right-of-ways, mechanical treatments, or planted stands of white pine.

Avian Sampling

Avian sampling was conducted from 19 May through the middle of July in 2011 and 2012 using 10 minute, 50-meter fixed radius point counts (Fig. 2). All birds detected by sight or sound were recorded. However, any birds heard or seen outside or flying over the point count circle were not used during the analysis. The distance to each bird was estimated and

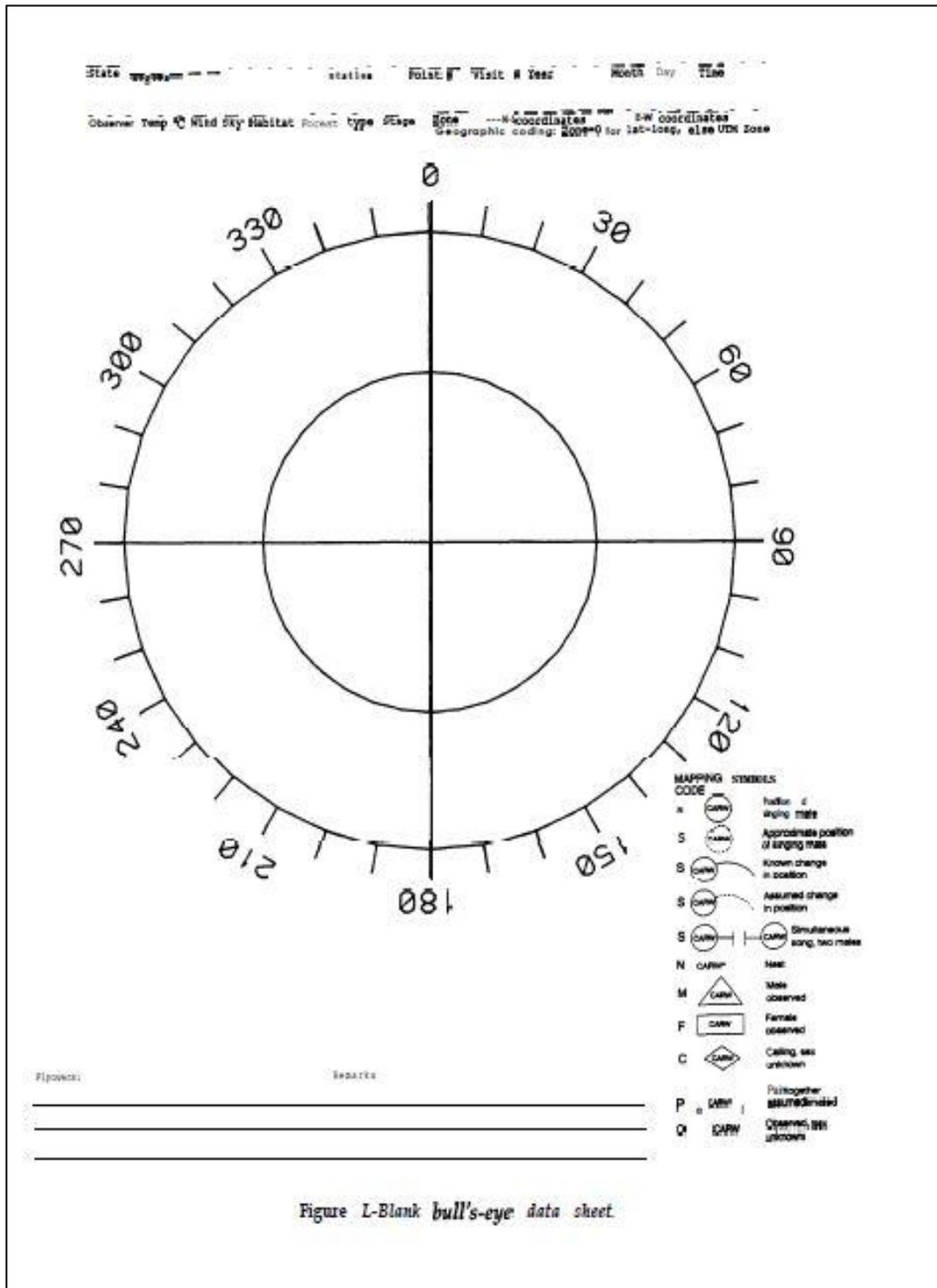


Figure 2. 50-meter fixed radius point-count data sheet (Hammel et al. 1996)

the age and sex of each individual were recorded when possible. Slope position at time of detection for each individual sampled was also noted if possible. All surveys were conducted from sunrise until 1000h (EST) on mornings with no precipitation and low winds (< 20 kph) (Ralph et al. 2003). Other variables recorded throughout sampling include time of survey, cloud cover, and temperature.

All points within the treatment and control areas were sampled two different times during the breeding season of 2011 and three different times during the breeding season of 2012. To minimize temporal variation between sampling efforts, the order in which a set of points was sampled was rotated between return visits within a given year and the overall order in which points were sampled was rotated between years. Sampling of the treatment and control sites were alternated daily in both 2011 and 2012.

Plot Selection for Vegetation Sampling

A subset of the total number of point locations established for avian sampling was used to sample vegetation for different site types as a result of logistical constraints. A total of 60 point count locations were chosen to measure vegetation with 30 points established in the treatment and control zones respectively. The treatment areas were subdivided into two blocks, each containing 15 vegetation plot locations. Conditions for exclusion were the same as those set for avian sampling and only those points that surveyed the targeted habitat types were selected for vegetation sampling. One 0.04-ha plot, 20 m x 20 m in dimensions, was centered around each of the 60 selected sampling points (James and Shugart 1970, Hamel et al. 1996). Sampling

occurred from May to July 2012 at which time landform, tree, and ground flora were measured in each plot.

Elevation (m), slope gradient (%), and slope aspect ($^{\circ}$) for each plot was derived from Digital elevation models (DEMs) and topographic maps provided by the SCDNR. All live trees > 10.2 cm DBH (Abella 2004) within the 0.04-ha plot were counted, identified, and measured (DBH and crown diameter). Average canopy height, measured with a clinometer, and age were determined for the stand from the measurements of a selected number (>4) of dominant and/or co-dominant trees within the plot. The dominant pine species and/or the dominant hardwood species were also noted as well as the number of snags and percentage of standing pine occupancy. Percent canopy cover was measured using a densitometer which was calculated from four measurements taken in each cardinal direction. Basal area and stem density were calculated from the collected data as well. At the center of each 20 m x 20 m plot, a 10 m x 10 m plot was nested to inventory ground flora (including tree seedlings and saplings < 10.2 cm in diameter) and to visually estimate percent cover of ground, shrub and herbaceous species. Each set of measurements were summarized and averaged when applicable to obtain the habitat variable values used in analyses mentioned later.

Avian/Vegetation Community Analyses

Relevant statistical analysis assumption checks, including the assumptions of normality and homogeneity of variances were examined. The assumption of normality was considered using graphical displays and the Kolmogorov-Smirnov D statistic while the homogeneity of variance assumption was checked with residual plots and Levene's Test (SAS Institute, 2012).

Differences in avian communities between the treatment and control zones were ascertained by the calculation of four descriptive indices. These indices include the total number of individual birds observed at each point (TOTAL) and species richness (S), the total number of species observed at each point. Species diversity (H') was also calculated using the Shannon-Weiner index,

$$H' = -\sum p_i \ln p_i$$

where p_i is the proportion of individuals found in the i th species and is estimated as n_i / N and evenness (E), from the equation,

$$E = H' / \ln S$$

(Magurran, 1988). The Sorensen's quantitative C_s index was used to calculate similarities in species composition between ecosystems based on the number of species present in each sample (Sorenson 1948). The index was calculated by the equation,

$$C_s = 2j / a + b$$

where j is the number of species common to sites 1 and 2, a is the number of species present in site 1 and b is the number of species present in site 2. The greater the index value, the more similarities the two sites share in terms of species compositions. Mean abundance of each species was also calculated by point and year for the unburned control sites and burned treatment sites as the total number of individuals of species x observed divided by the total number of individuals of all species observed. This number was then multiplied by the number of times each point was surveyed to determine abundance by year.

A Kolmogorov test was used as the sample size exceeds 2,000, conditions under which a Shapiro Wilk test is less reliable (SAS Institute, 2012). As a result of variability common to point count data, the assumption of normality was frequently violated. Consequently, nonparametric statistical methods were used for several of the analyses.

For the indices of species diversity, total number of individual birds observed at each point, and species richness, a mean was calculated for the treatment and control sites and used as the dependent variable in sign-rank comparisons made to detect significant differences between the two site types (PROC NPAR1WAY WILCOXON; SAS institute 2012). Tests were performed on each sampled year individually and both years combined. A significance level (α) of 0.10 was used for all analyses.

Specific associations between habitat characteristics and focal species were also explored. Seven focal species were selected for individual analysis. Only data for 2012 were considered for this analyses as vegetation sampling was limited to that year. The selection criteria included Partners in Flight (PIF) priority scoring for the Southern Blue Ridge Physiographic area (Hunter et al. 1999), sample size ($n > 9$), nesting guild classification (ground/shrub, canopy, cavity), and foraging guild classification (ground, foliage, bark, hawk). Presence or absence of each focal species was noted for each point in each year. A correlation analysis for 13 habitat variables (Fig. 3) was conducted (PROC CORR; SAS Institute 2012). Variables with a high correlation to one or more of the other independent variables (Pearson's $r > 0.80$) were excluded from the cluster analysis. Cluster analysis (K-means method) was then performed to examine selected habitat characteristics for focal species based on Euclidean distance between mean values of habitat variables (PROC CLUSTER; SAS Institute 2012).

To aid in comprehension of avian-habitat relationships, logistic regression with a generalized linear mixed model was used to consider habitat variables potentially relevant for predicting the presence or absence of each focal species (PROC GLIMMIX; SAS Institute 2012). Its use in this study was to draw a random intercept separately and independently for point within each round of year two of the study, thereby accounting for the variation of each sampling event in that year.

Similar to cluster analysis, only data for 2012 were considered for logistic regression as vegetation sampling was limited to that year. The results from the correlation analysis (PROC CORR; SAS Institute 2012) previously conducted were applied to the regression model as well. Variables with a high correlation to one or more of the other independent variables (Pearson's $r > 0.80$) were excluded from the regression analysis. A significance level of 0.15 was set for a variable to enter into and/or remain in the model and to ensure all influential habitat variables were identified.

To investigate differences in habitat selection between species groups, all species detected by point counts were categorized by their respective foraging and nesting guilds (Hamel et al. 1992). Foraging guilds are comprised of hawkers, foliage-gleaners, ground-gleaners, and bark-gleaners. Nesting guilds are comprised of canopy nesters, cavity nesters, and ground or shrub nesters. Foraging or nesting guilds encountered only once were excluded from the analysis. Frequencies of individual foraging guilds in the treatment and control areas were determined and a Chi-square test (PROC FREQ; SAS institute 2012) was conducted to examine differences in foraging guild structure among site types. Similar analysis was performed to examine differences in nesting guild structure among site types. For guilds exhibiting significant differences between site types ($p < 0.10$), Bonferroni's adjustment (α/n) for simultaneous

hypothesis tests was used to determine which treatment type was preferred or avoided by individual foraging ($n = 4$) and nesting guilds ($n=3$).

Table 1. Descriptions and sampling methodology of the 13 environmental variables surveyed.

Abbreviation	Description	Method
Elev	Elevation from sea level (m)	Geographic information systems
Slope	Percent slope	Geographic information systems
Basal	Total basal area (m ²)	Basal area prism
Crown	Mean of crown diameter (m)	Logger's tape
DBH	Mean diameter of trees > 10.2 cm DBH	Logger's tape
Pine	Percent coniferous coverage	Count
Ntrees	Total number of trees	Count
Nsnags	Total number of standing dead trees	Count
Height	Mean height of 5 selected canopy trees	Clinometer
Cancov	Percent canopy coverage	Densitometer
Herbcov	Percent herbaceous coverage	Visual estimate
Shrubcov	Percent shrub coverage	Visual estimate
Groundcov	Percent understory coverage	Visual estimate

Visualization

To begin the visualization process, the first objective was to virtually recreate the forested ridge-top environments of the Jocassee Gorges. The elevation for the landscape were obtained from a 10 meter digital elevation map (DEM) which was imported into Visual Nature Studio (VNS), a software package specialized in creating photorealistic images and animations of forested and urban terrains among many others. To visualize the forest vegetation, user-made tree models were used instead of models included with VNS to increase realism and to ensure that more site-specific tree forms were employed throughout visualization. The tree models were made from photographs of tree species local to the Jocassee Gorges taken with a digital camera during the summer of 2012. Once loaded into Adobe Photoshop, the photographs were edited to remove surrounding vegetation leaving the tree of interest alone in the foreground while the background of the photo was painted black. The images were saved as JPG (JPEG, Joint Photographic Experts Group, compressed image) files and imported into the VNS graphics library to use as models.

To realistically visualize the forest environment, tree models were connected to actual forest structure in terms of characteristics such as tree height and density. This was performed by employing forest inventory data and a geo-referenced landcover map. Forest inventory data from test plots included tree height, species, diameter at breast height (DBH), basal area, and density. An analysis of the data was conducted to determine the mean and standard deviations for tree height and DBH in each plot. The landcover map was then imported into ArcMap where each map symbology value was given a different RGB (red, green, blue) color code and then exported as a GeoTIFF file (a Tagged Image File Format image retaining its spatial coordinates).

When imported into VNS, the GeoTIFF file acted as a color map to visualize varying forest ecosystems of different structure based upon the landcover value.

Using the ecosystem function in VNS, ridge-top forest “ecosystems” were constructed. An ecosystem is defined in VNS as an association of plant species all sharing common characteristics such as height, density, and relative frequency. Appropriate tree models were placed in the canopy and understory layers based on species from the forest inventory data. Average height was used for the main tree height while the standard deviation was used as an offset factor to vary tree height. An included ground texture representing a forest floor with leaf litter was assigned to represent the ground.

To add realism and increase aesthetical appeal, certain features were added to the initial environment. These features included sky and cloud models, lighting, and shadows. Most models were added from VNS’ built in library of objects with minimal editing.

Once the initial virtual forest environment was created, models were applied to the forest data to determine changes with future succession and growth. The data from each model was then applied to the initial visualization to convey changes in forest stand structure and composition that occur due to fire disturbance events. The final visualization of long term fire management of ridge-tops was adapted from Brose and Waldrop (2006). Animations were then produced using the animating editor included in the VNS package. The animation editor functions as a sequence moving through key frame images. Each key frame image is rendered in sequence as a still frame with VNS adding a transition between each frame. When combined, it produces smooth animations through the entire sequence.

CHAPTER III

RESULTS

Avian Sampling

Community Indices

During the 2011-2012 breeding seasons, 6,263 individuals of 59 species were detected during point counts with 2,634 individuals of 54 species in 2011 and 3,629 individuals of 54 species in 2012. Based on Wilcoxon Rank Sum tests, significant differences between treatment and control sites were detected in species diversity ($Z = 3.36$, $p = .0008$), species richness ($Z = 3.35$, $p = .0008$), and total number of individuals per point ($Z = 3.14$, $p = .0017$) in 2012. When data were combined across years, differences were detected for species diversity ($Z = -3.3$, $p = .0010$), species richness ($Z = -3.27$, $p = .0011$), and total individuals ($Z = -3.16$, $p = .0016$). No differences were detected for any of the four indices in 2011, and no differences were detected for evenness among individual years or between combined years.

When comparing indices for plots that were burned in 2010 and plots that were burned in 2011, only data collected in 2011 provided evidence of differences. Differences were detected in species diversity ($Z = 1.4$, $p = .0810$), species richness ($Z = 1.42$, $p = .0782$), and total number of individuals ($Z = 1.33$, $p = .0923$) in 2011. No other differences in indices were detected between burn plots for the year of 2012 or across years combined.

For the year of 2012, it was noted that all indices compared between the treatment and control sites in which significant differences were detected (diversity, richness, and total) expressed

Table 2. Average (standard deviation) species diversity, evenness, richness, and total number of birds per plot for the burned treatment plots and the control plots for individual years surveyed and across years combined.

	Burn 2011	Control 2011	Burn 2012	Control 2012	Burn Years Combined	Control Years Combined
Diversity (H')	1.66 (0.56) a	1.55 (0.7) a	1.27 (0.58) a	1.10 (0.6) b	1.34 (0.64) a	1.16 (0.72) b
Evenness (E')	0.986 (0.02) a	0.986 (.02) a	0.989 (0.02) a	0.988 (.02) a	0.986 (0.02) a	0.985 (0.02) a
Richness (S)	6.18 (2.99) a	6.12 (3.57) a	4.30 (2.67) a	3.60 (2.22) b	5.31 (2.79) a	4.86 (3.14) b
Total (TOTAL)	6.90 (3.48) a	6.90 (4.23) a	4.72 (2.68) a	3.95 (2.51) b	5.81 (3.26) a	5.36 (3.65) b

*Values with different lower case letters for each pair within rows are significantly different ($p < 0.10$).

Table 3. Average (standard deviation) species diversity, evenness, richness, and total number of birds per plot for plots that were burned in 2011 and plots that were burned in 2012 for individual years surveyed and across years combined.

	Plots Burned in 2010	Plots Burned in 2011	Plots Burned in 2010	Plots Burned in 2011	Plots Burned in 2010	Plots Burned in 2011
	2011	2011	2012	2012	Years Combined	Years Combined
Diversity (H')	1.62 (0.51) a	1.72 (0.59) b	1.27 (0.61) a	1.26 (0.62) a	1.44 (0.61) a	1.49 (0.67) a
Evenness (E')	0.984 (.02) a	0.989 (0.01) a	0.987 (0.02) a	0.99 (0.02) a	0.986 (0.02) a	0.99 (0.02) a
Richness (S)	5.81 (2.36) a	6.79 (3.57) b	4.34 (2.19) a	4.27 (2.21) a	4.64 (2.40) a	4.9 (3.25) a
Total (TOTAL)	6.53 (2.80) a	7.5 (4.12) b	4.76 (2.54) a	4.66 (2.65) a	5.15 (2.83) a	5.38 (3.78) a

*Values with different lower case letters for each pair within rows are significantly different ($p < 0.10$).

notably higher values for the burned sites. Similar results were observed across years combined in that all differences in indices were also higher for the burned sites than the control sites. For the year of 2011, comparisons of indices between plots burned in 2010 and plots burned in 2011 revealed that species diversity, species richness, and total number of individuals per point was higher for plots burned in 2011.

Concerning the treatment and control sites at the Gorges, comparisons of species similarities through the Sorensen's quantitative C_s showed that species data across years combined were the most similar ($C_s = 8.041$). The data that showed the least similar results was that of 2012 ($C_s = .7586$). The data for 2011 returned a marginally higher quantitative C_s than the data for 2012 ($C_s = .7674$). When considering the plots burned in 2010 and plots burned in 2011, the highest similarities were observed in 2012 ($C_s = .8718$), while the lowest similarities were observed in 2011 ($C_s = .8537$). When combining data across years, the Sorensen's quantitative value fell in between those of individual years ($C_s = .8696$). Results for the abundance index are summarized in Table 4.

Table 4. Mean abundance (standard deviation) of species detected in spring 2011 and 2012 by 50 m radius point counts at the Jocassee Gorges, South Carolina.

Species	Burn		Control	
	2011	2012	2011	2012
Yellow-billed cuckoo <i>Coccyzus americanus</i>	0.17 (0.07)	0.25 (0.1)	0.18 (.08)	0.36 (0.22)
Worm-eating warbler <i>Helmitheros vermivorous</i>	0.18 (0.11)	0.63 (0.45)	0.16 (.13)	0.5 (0.35)
White-breasted nuthatch <i>Sitta carolinensis</i>	0.17 (0.11)	0.62 (0.52)	.16 (.07)	0.83 (0.85)
Tufted titmouse <i>Baeolophus bicolor</i>	0.29 (0.14)	0.72 (0.55)	.3 (.15)	0.85 (0.62)
Scarlet tanager <i>Piranga olivacea</i>	0.32 (0.14)	0.87 (0.77)	.3 (.17)	0.94 (0.82)
Red-tailed hawk <i>Buteo jamaicensis</i>	0.15 (0.03)			
Ruby-throated hummingbird <i>Archilochus colubris</i>		0.28 (0.11)		
Red-eyed vireo <i>Vireo olivaceus</i>	0.34 (0.26)	0.84 (0.61)	0.34 (0.19)	1.14 (.85)
Red-bellied woodpecker <i>Melanerpes carolinus</i>	0.14 (0.02)	0.26 (0.13)	0.13 (0.04)	0.31 (0.05)
Pine warbler <i>Setophaga pinus</i>	0.14 (0.04)	0.33 (0.19)	0.21 (0.11)	0.82 (0.9)
Pileated woodpecker <i>Dryocopus pileatus</i>	0.17 (0.06)	0.23 (0.05)	0.15 (0.07)	0.31 (0.09)
Ovenbird <i>Seiurus aurocapillus</i>	0.16 (0.03)	0.36 (0.19)	0.19 (0.06)	0.3 (0.09)
Northern cardinal <i>Cardinalis cardinalis</i>	0.22 (0.13)	0.3 (0.14)	0.23 (0.1)	0.35 (0.21)
Mourning dove <i>Zenaida macroura</i>	0.17 (0.11)	0.22 (0.06)	0.18 (0.06)	0.59 (0.29)
Indigo bunting <i>Passerina cyanea</i>	0.43 (0.26)	1.04 (0.79)	0.28 (0.21)	0.99 (0.91)
Hooded warbler <i>Wilsonia citrine</i>	0.17 (.08)	0.3 (0.15)	0.2 (0.14)	0.49 (0.49)
Hairy woodpecker <i>Picoides pubescens</i>	0.18 (.07)	0.39 (0.31)	0.14 (0.04)	0.88 (0.4)
Great crested flycatcher <i>Myiarchus crinitus</i>	0.15 (.04)	0.29 (0.21)	0.15 (0.04)	
Eastern wood-pewee <i>Contopus virens</i>	0.16 (0.06)	0.33 (0.37)		0.28 (0.12)
Eastern towhee <i>Pipilo erythrophthalmus</i>	0.17 (0.08)	0.26 (0.11)	0.19 (0.1)	0.34 (0.15)
Eastern phoebe <i>Sayornis phoebe</i>	0.13 (0.03)			
Downy woodpecker <i>Picoides pubescens</i>	0.25 (0.14)	0.63 (0.41)	0.17 (0.07)	0.54 (0.36)
Common raven <i>Corvus corax</i>	0.11 (0.02)			
Carolina wren <i>Thryothorus ludovicianus</i>	0.29 (0.16)	0.64 (0.54)	0.3 (0.16)	0.67 (0.74)
Carolina chickadee <i>Parus carolinensis</i>	0.2 (0.12)	0.87 (0.74)	0.2 (0.09)	0.72 (0.59)
Brown thrasher <i>Toxostoma rufum</i>	0.17 (0.06)			
Blue jay <i>Cyanocitta cristata</i>	0.28 (0.39)	0.43 (0.44)	0.18 (0.08)	0.48 (0.36)
Blue-headed vireo <i>Vireo solitarius</i>	0.29 (0.18)	1.02 (0.63)	0.32 (0.19)	1.22 (.89)
Blue-gray gnatcatcher <i>Poliaoptila caerulea</i>	0.27 (0.18)	0.22 (0.08)	0.15 (0.05)	0.37 (0.19)
Black-throated green warbler <i>Dendroica virens</i>	0.32 (0.35)	0.71 (0.47)	0.3 (0.18)	10.06 (0.88)
Black-and-white warbler <i>Mniotilta varia</i>	0.55 (1.09)	0.25 (0.09)	0.14 (0.03)	0.32 (0.14)
American goldfinch <i>Carduelis tristis</i>		0.33 (0.24)		0.47 (0.32)
Acadian flycatcher <i>Empidonax vireescens</i>		0.23 (0.07)	0.16 (0.07)	
American crow <i>Corvus brachyrhynchos</i>	0.16 (0.08)	0.28 (0.1)	0.15 (0.07)	0.31 (0.17)

(Blank fields represent years in which a species was not detected in sufficient number to calculate abundance)

Cluster analysis

Cluster analyses were performed to explore species composition and association within the different habitat types but results were inconclusive. The preceding correlation analysis revealed the number of trees to be highly correlated with basal area ($r = 0.8077$, $p = <.0001$). The number of trees variable was excluded from the analysis as it also had a relatively high, although insignificant, correlation with other variables. It was also found that ground cover was highly correlated to shrub cover ($r = 0.9496$, $p = <.0001$). As a result, ground cover was also removed from the analysis as it had a marginally high correlation with other variables as well. Two separate clusters were identified. However, the only variable that recognizably warrants a division into two clusters is altitude, which has a noticeably higher mean value in the second cluster (i.e., mean altitude of 534.19 m for the first cluster and mean altitude of 676.57 m for the second cluster). Although priority species occur more frequently in cluster two of the two clusters, the lack of clear discrimination in habitat characteristics may be a result of similar proportions of non-priority species within each cluster. Limited size of the study area, juxtaposition of habitat types, and the variable ranges of priority species are other possible reasons that patterns between habitat preferences for priority and non-priority species were not detected.

Logistic Regression

A subset of 11 of the original 13 habitat characteristics measured was included in the logistic regression model to identify important variables related to bird occurrence. Two of the habitat variables (N trees and ground cover) were removed following correlation analysis ($r >$

0.8). Generalized linear mixed models were created for the seven priority species. Scarlet tanagers were not found to have significant relationships to habitat characteristics. Caution should be exercised in that the following analyses are based on relatively small and localized sample sizes. The information highlights important habitat characteristics related to priority species occurrence but conclusions are only exploratory.

A model was created for presence or absence of priority species. The priority species model included altitude ($F = 5.39$, $p = .0216$), slope gradient ($F = 2.99$, $p = .0855$), and diameter at breast height ($F = 2.36$, $p = 0.1264$). The treatment effect between the areas treated with prescribed fire and the controlled unburned areas ($F = 1.83$, $p = 0.1784$) was only marginally insignificant at $\alpha = .15$, but was consequently excluded from the model.

Concerning individual priority species, the important habitat characteristics associated with Eastern wood pewee occurrence were slope gradient ($F = 2.5$, $p = 0.116$) and percent pine ($F = 2.38$, $p = 0.1248$). Herbaceous cover ($F = 2.04$, $p = 0.1549$), although just marginally insignificant was excluded from the model. For ovenbirds, habitat characteristics potentially useful for predicting occurrence were found to be diameter at breast height ($F = 3.29$, $p = 0.0715$) and canopy height ($F = 2.66$, $p = 0.105$).

The model for worm-eating warblers included shrub cover ($F = 2.46$, $p = .1187$) and percent pine ($F = 2.12$, $p = .1477$). Gradient ($F = 1.96$, $p = .1633$) was also a notable characteristic potentially associated with worm-eating warbler occurrence but was not significant when compared to the set alpha level for logistic regression analyses. Significant habitat characteristics found to be related to hooded warbler occurrence were the treatment effect between areas treated with prescribed fire and control areas ($F = 4.88$, $p = 0.0286$), diameter at

breast height ($F = 2.34$, $p = 0.1280$), and crown diameter ($F = 2.91$, $p = 0.0899$). Canopy cover ($F = 1.36$, $p = 0.2461$) exhibited some relation to occurrence but was not included in the model.

The black and white warbler model included two significant variables related to occurrence, number of snags ($F = 2.39$, $p = 0.1238$) and canopy cover ($F = 2.22$, $p = 0.1386$). The crown diameter ($F = 1.53$, $p = 0.2172$) and canopy height ($F = 1.65$, $p = 0.2006$) variables were marginally insignificant but notable. The model created for black throated green warblers included three habitat variables. Areas treated with prescribed fire and control areas ($F = 3.44$, $p = .0656$), altitude ($F = 3.53$, $p = .0622$), and percent pine ($F = 4.52$, $p = .035$) were all significant habitat characteristics.

Foraging and Nesting Guilds

Overall comparisons of observed versus expected values and mean frequencies, or the average rate at which guild members occurred during sampling for foraging guilds among areas treated with prescribed fire and control areas found significant differences for years combined ($X^2 = 21.4993$, $p = 0.0003$) and for the year of 2012 ($X^2 = 20.6074$, $p = 0.0004$), but no significant difference between sites for the year of 2011. When considering individual foraging guilds, significant differences were found across years combined for foliage-gleaners ($X^2 = 7.3215$, $p = 0.0068$) which occurred more frequently in the control areas, and hawkers ($X^2 = 10.8694$, $p = 0.0010$) which occurred more frequently in the burned areas. No significant differences were found for ground-foragers or bark-gleaners across years combined. Similarly, for the year of 2012, significant differences were found for foliage-gleaners ($X^2 = 8.0297$, $p = 0.0046$) which occurred more frequently in the control areas, and hawkers ($X^2 = 12.3061$, $p =$

0.0005) which occurred more frequently in the burned areas. Significant differences were not detected for ground-foragers or bark-gleaners in that year. No significant differences were detected for individual foraging guilds for the year of 2011.

Significant differences between areas treated with prescribed fire and control areas for nesting guilds were found when years were combined ($X^2 = 10.267$, $p = 0.0164$) and for the year of 2012 ($X^2 = 8.9605$, $p = 0.0298$), but no overall significant difference in nesting guilds for the year of 2011. When considering individual nesting guilds, significant differences were found in favor of the control areas for canopy nesters ($X^2 = 10.0856$, $p = 0.0015$) across years combined, with no significant differences detected for cavity or ground/shrub nesters. For the year of 2012, significant differences were detected for canopy nesters ($X^2 = 8.8377$, $p = 0.0030$) which favored the unburned areas, with no significant differences detected for cavity nesters or ground/shrub nesters. No significant differences were detected for individual nesting guilds for the year of 2011.

Significant differences between plots that were burned in 2010 and plots that were burned in 2011 for foraging guilds were found when years were combined ($X^2 = 29.9060$, $p < 0.0001$), for the year of 2011 ($X^2 = 15.2619$, $p = 0.0042$), and for the year of 2012 ($X^2 = 18.0620$, $p = 0.0012$). When considering individual foraging guilds across years, significant differences were detected for foliage-gleaners ($X^2 = 23.2558$, $p < 0.0001$) which occurred more frequently in plots burned in 2010, and also bark-gleaners ($X^2 = 12.1169$, $p = 0.0005$), and hawkers ($X^2 = 8.6960$, $p = 0.0032$) which preferred plots burned in 2011. No significant differences were found for ground foragers for years combined. For the year of 2011, significant differences were detected for foliage-gleaners ($X^2 = 10.2959$, $p = 0.0013$) which were more common in areas burned in 2010. No differences were detected for bark-gleaners, hawkers, or ground-foragers for the year

of 2011. For the year of 2012, significant differences were detected for foliage-gleaners ($X^2 = 13.3216$, $p = 0.0003$), bark-gleaners ($X^2 = 8.7559$, $p = 0.0031$), and hawks ($X^2 = 6.6015$, $p = 0.0102$) with each guild exhibiting similar preference to area type as was found when examining data for years combined. No significant differences were detected for ground-foragers for the year of 2012.

Significant differences between plots that were burned in 2010 and plots that were burned in 2011 for nesting guilds were found when years were combined ($X^2 = 32.8356$, $p < 0.0001$), for the year of 2011 ($X^2 = 14.8637$, $p = 0.0019$) and for the year of 2012 ($X^2 = 19.1110$, $p = 0.0003$). When considering individual nesting guilds, significant differences were detected for canopy ($X^2 = 12.7051$, $p = 0.0004$) and cavity ($X^2 = 25.2682$, $p < 0.0001$) nesters for years combined. For the year of 2011, differences were detected for canopy ($X^2 = 4.9677$, $p = 0.0258$) and cavity ($X^2 = 11.3525$, $p = 0.0008$) nesters. Significant differences were detected for canopy ($X^2 = 8.0149$, $p = 0.0046$) and cavity ($X^2 = 14.9659$, $p = 0.0001$) nesters for the year of 2012. In each comparison of nesting guild frequency between plots that were burned in 2010 and plots that were burned in 2011, canopy nesters exhibited a significantly higher rate of detection in plots burned in 2010 while cavity nesters exhibited a significantly higher rate of detection in plots burned in 2011. No significant differences were detected for ground/shrub nesters for years combined or for the individual years of 2011 and 2012.

Table 5. Statistical differences observed for foraging guild structure for the burned treatment plots and the control plots for individual years surveyed and across years combined. Blank fields represent no statistical difference between site types.

	2011	2012	Years Combined
Overall Structure between sites		X ² = 20.6074, p = .0004	X ² = 21.4993, p = .0003
Foliage-gleaners		X ² = 8.0297, p = .0046	X ² = 7.3215, p = .0068
Bark-gleaners			
Ground-gleaners			
Hawkers *		X ² = 12.3061, p = .0005	X ² = 10.8694, p = .0010

* This symbol represents guilds associated with burned treatment sites

Table 6. Statistical differences observed for foraging guild structure between plots burned in 2010 and plots burned in 2011 for individual years surveyed and across years combined. Blank fields represent no statistical difference between site types.

	2011	2012	Years Combined
Overall Structure between sites	X ² = 15.2619, p = 0.0042	X ² = 18.0620, p = 0.0012	X ² = 29.906, p < 0.0001
Foliage-gleaners	X ² = 10.2959, p = 0.0013	X ² = 8.0297, p = 0.0046	X ² = 7.3215, p = 0.0068
Bark-gleaners *		X ² = 8.7559, p = 0.0031	X ² = 12.1169, p = 0.0005
Ground-gleaners			
Hawkers *		X ² = 6.6015, p = 0.0102	X ² = 8.6960, p = 0.0032

* This symbol represents guilds associated with plots burned in 2011.

Table 7. Statistical differences observed for nesting guild structure for the burned treatment plots and the control plots for individual years surveyed and across years combined. Blank fields represent no statistical difference between site types.

	2011	2012	Years Combined
Overall Structure between sites		X ² = 8.9605, p = 0.0298	X ² = 10.267, p = 0.0164
Canopy nesters		X ² = 8.8377, p = 0.0030	X ² = 10.0856, p = 0.0015
Cavity Nesters			
Ground/Shrub Nesters			

* This symbol represents guilds associated with burned treatment sites.

Table 8. Statistical differences observed for nesting guild structure between plots burned in 2010 and plots burned in 2011 for individual years surveyed and across years combined. Blank fields represent no statistical difference between site types.

	2011	2012	Years Combined
Overall Structure between sites	X ² = 14.8637, p = 0.0019	X ² = 19.1110, p = 0.0003	X ² = 32.8356, p < 0.0001
Canopy nesters	X ² = 4.9677, p = 0.0258	X ² = 8.0149, p = 0.0046	X ² = 12.7051, p = 0.0004
Cavity Nesters *	X ² = 11.3525, p = 0.0008	X ² = 14.9659, p = 0.0001	X ² = 14.9659, p = 0.0001
Ground/Shrub Nesters			

* This symbol represents guilds associated with plots burned in 2011.

Visualization

The results of this section will be expressed as figures produced via visualization software. The utility of the software will be discussed in the following chapter.



Figure 3. Visualization of typical present-day ridge top forest stand at The Jocassee Gorges characterized by a



Figure 4. 4. Visualization of ridge top forest stand immediately following low severity fire at the Jocassee Gorges characterized by newly created snags and blackened images representing the impact of low-severity fire on the shrub layer .



Figure 5. Visualization of ridge top forest stand 2 years after low severity fire at the Jocassee Gorges characterized by an increase in snag density/canopy gaps as a result of delayed tree mortality and regeneration of the shrub layer.



Figure 6. Visualization of long term fire management (Initial high severity fire followed with maintenance of frequent (2 year cycle) low severity fires) of ridge-tops at the Jocassee Gorges characterized by a drastic reduction of hardwood occupancy with an absence of mid-story and moderately dense to dense grass/shrub layer (Image adapted from Brose and Waldrop 2006). Habitats of this type in the SC Mountains are expected to support such species as bachman's sparrows (*Aimophila aestivalis*), bobwhite quail (*Colinus virginianus*), prairie warblers (*Dendroica discolor*), ruffed grouse (*Bonasa umbellus*) and potentially red-cockaded woodpeckers (*Picoides borealis*) among many others.

CHAPTER IV

DISCUSSION

Community Indices

When considering the following, it should be noted that there is some limitation stemming from the lack of replication of varying burn conditions. For the 2012 sampling year and both sampling years combined, the burned sites surveyed at the Jocassee Gorges are represented by statistically higher values for the community indices of species diversity, species richness, and total numbers of species per point sampled than are the control sites. These findings coincide with other studies conducted in the Blue Ridge and Appalachian mountains that report higher bird diversity, richness, and density in sites that were disturbed by management activities compared to undisturbed forests (Annand and Thompson 1997, Baker and Lacki 1997). Weakland et al. (2002) also reported that the mosaic of shrub cover and tree mortality created by patch burns promoted higher bird species richness in West Virginia, while Zebehazy et al. (2004) found an increase in richness and abundance resulting in part from fuel reduction burns in the Piedmont region of South Carolina.

Vegetation structure has strong impacts on the diversity and composition of avian communities. Disturbance by fire has the potential to enhance diversity on stand and landscape levels by altering the vertical strata and horizontal distribution of vegetation and by staggering successional stages, providing an array of foraging and nesting opportunities (Askins 2000, Brawn et al. 2001). Increase in habitat diversity and structure resulting from fire management are possible explanations for higher index values for the burned sites in relation to the control sites. Typically, ridge-top pine forests of the Blue Ridge Mountains are xeric forests which were

historically dominated by shade-intolerant pines overtopping a moderate to high density of shrub and herbaceous species in the understory. Historic fire regimes and poor site conditions often deterred the establishment of competing hardwoods and shade tolerant understory species. Human disturbance and fire exclusion have led to increased homogeneity across the landscape, diluting transitional zones and edge habitat capable of supporting high bird diversities.

Although index values of avian diversity and richness for the 2011 survey year are higher in the burned sites than in the control sites, they are not statistically different. The reason for the similarities in composition between treatment and control for the year of 2011 is difficult to discern. It has been reported that bird response to fire varies according to fire severity and the corresponding post-burn conditions (Hejl 1994). Studies on fire severity in relation to avian community response have found that low intensity burns with little subsequent tree mortality have little detectable effect on many bird species or community parameters (Greenberg et al. 2007, Artman et al. 2001).

Tree mortality eventually results in canopy gaps, allowing sunlight to penetrate to the forest floor with less resistance. This new light source promotes the growth of previously dormant or suppressed vegetation, providing potential for new growth and greater structural variation within a stand. Among the few stands affected by high severity fire where many trees were killed outright, radical differences in structure were observed when compared to control areas. The most notable distinctions included diminished over-story retaining pine dominants and hardwood snags with a dense shrub layer of *Vaccinium pallidum* (low-bush blueberry) and the near absence of species in the mid-story. Stands impacted by high severity fires also supported higher densities of regenerating yellow pine species than did control stands and stands impacted by low severity fires. Mostly low to moderate burn severities within treatment sites at

the Jocassee are one explanation for similarities in habitat structure and the delay in avian community response.

Another explanation for the closeness of the index values observed between treatment and control could be a relatively even shift in bird numbers as a result of altered foraging and nesting resources following controlled burning. Certain species, such as the indigo bunting and eastern wood peewee, were detected in greater number in burned areas while species such as the hooded warbler and black throated green warbler were detected in greater number in control areas. Although there are notable species differences between treatment and control, the number of individuals present during sampling in both the treatment and control are relatively similar. The majority of the remaining species present throughout surveying for the year of 2011 were seemingly less affected by the controlled burn treatment.

As for the separation in index values between the treatment and control for the sampling year of 2012, multiple studies have been conducted on the response of biotic communities in the years following fire. The full effects of fire are not often realized immediately following disturbance events. A recent study produced results showing that avian species richness, abundance, and conservation value are low for a given time after fire (Pons and Clavero, 2009). These index values, however, were found to increase as time since fire increased. Lags in avian response may be due to delayed habitat changes, such as the fall of dead hardwoods in years after fire or increases in shrub cover. A similar study on post-fire bird response supports these findings, suggesting that tree mortality rate and bird abundance tended to increase with time since fire (Lowe et al. 2009).

Upon observing index values comparing plots that were burned in 2010 and plots that were burned in 2011, it was found that only data for the 2011 sampling year produced statistical differences in avian community composition between burn plots. Tests for index values of species richness, species diversity, and total number of species returned higher statistical values for plots burned in 2011 within that survey year alone. Interpretation of these results can be difficult based on past conflicting research results including studies suggesting that observed values are low after fire (Pons and Clavero 2009, Lowe et al. 2009) and those that show an increase in diversity and number following fire (Kirkpatrick et al. 2006, Koivula and Schmiegelow 2007, Kotliar et al. 2007, Dickson et al., 2009). One might expect avian productivity to be higher in areas with a greater time since fire when considering recovery of the habitat including shrub and grass-forb cover increases typically occurring from the first to the second year after fire events (Smucker et al. 2005). However, there are multiple drivers and characteristics of individual burns which play significant roles in determining the ultimate effects of fire on the impacted ecosystem.

In addition to general responses of species to fire, represented by the overall frequency of occurrence, species may respond unexpectedly to particular fire events. Watson et al. (2012) found that sites where patchy vegetation remained post-fire supported more species than uniformly burned sites within the first 5 years post-fire. These patchy fires may result in faster recovery of avifaunal communities, and provide habitat for some species that may be extirpated following more uniform fires.

Furthermore, nesting or foraging requirements do not necessarily restrict birds to specific habitat conditions. Ground nesting species, including ovenbirds and wood thrushes, have been frequently observed foraging within recently burned areas (Artman et al. 2001). Woinarski

(1990) suggests that fire can potentially enhance food resource availability for ground foraging birds by removing leaf litter and exposing insects and seeds. As both treatment plots were burned within one year of the other and at similar severities, on average, it is likely that the explanation for higher index values for plots burned in 2011 in that year is the new release and availability of foraging resources despite possibly unsuitable nesting conditions.

Similarity Indices

Results from the Sorensen's quantitative C_s , an index used to examine similarities between communities, showed that similarities between the burned areas and the control areas were greatest across years combined. This could indicate a shift in bird habitat usage between the first and the second year of sampling as a result of resource availability and vegetation recovery rates. A probable reason for this higher value in relation to values for individual sampling years may be contrariwise usage rates of particular species for treatment and control sites during those individual years. When data are combined across years, species that may not have been present in similarity analyses for an individual year and area type may then be present in the analyzed comprehensive data, thereby, increasing similarity value.

The data for 2012 returned the lowest similarity value between the burned and control areas. This value may represent a stronger shift in species usage towards burned habitats as time since fire increases and flora recovers as concluded in related studies (Pons and Clavero 2009, Lowe et al. 2009). Multiple species, including the ruby-throated hummingbird and great crested flycatcher, were only recorded in burned areas during sampling for the year of 2012.

Furthermore, statistical differences in total number of birds between treatment and control areas

were detected based on Wilcoxon Rank Sum tests, supporting the lower similarity value observed. Although higher than the similarity value for 2012, the value for 2011 is still low in comparison to that for combined years, indicating recognizable differences in avian community structure.

Upon examination of similarities between plots burned in 2010 and plots burned in 2011, the highest value was observed for 2012, with the lowest similarity existing between plots in 2011. This is a series of associations that could be expected given that habitat differences between burn plots were most likely at their greatest immediately following prescribed fire in 2011, as plots burned in 2010 had one full growing season to recover from fire while plots burned in 2011 had been burned only months before sampling in that year. This information condones a lower similarity value for 2011 and a higher value for 2012 as vegetation in both of the burn plots had time to regenerate after treatment and ability to support similar communities increased accordingly.

Cluster Analysis

The results of the exploratory cluster analysis are inconclusive. Although two individual clusters have been identified, the only variable separating said clusters is altitude. As a consequence of minimal division between clusters, reliable inference on discriminatory habitat characteristics conducive to the presence of priority species cannot be extracted from the results of the analysis. Although priority species occur more frequently in the second cluster of the two clusters, similar proportions of priority and non-priority species exist between them, further complicating interpretation of the analysis. Limited size of the study area, juxtaposition of

habitat types, and the variable ranges of priority species are other possible reasons that patterns between habitat preferences for priority and non-priority species were not detected.

Habitat Associations of Focal Species

Focal Species as a Group

The initial model created for all focal species as a group includes three important habitat variables: altitude, slope, and diameter at breast height. Increases in elevation can bring lower temperatures, increased precipitation, shorter growing seasons, and greater wind velocities (Shanks 1954). Some studies have found a steady decline of species richness with elevation, while others report a unimodal relationship, with a peak in species richness at low to mid elevations (Rahbek 1995, Stotz et al. 1996, Stotz 1998, Brown 2001, Lomolino 2001). In this case, the study site lies at the interface of the extreme southern section of the mid-elevation Blue Ridge physiographic province and the lower-elevation Piedmont region of South Carolina where altitude is likely a limiting factor related to bird occurrence as it often has association to habitat parameters. This region supports many species at the Southern ends of their range that tend to adhere to specific elevational gradients in the area, including the focal species black-throated green and black and white warblers, among others. The information provided justifies a positive relationship between increases in altitude and focal species.

As with elevation, slope can have meaningful impacts on biotic communities in landscapes with topographical variation. Long recognized as an important landscape variable, slope affects the amount and daily cycle of solar radiation received at different times of the year and has a strong influence on microclimate, humidity, and soil moisture, which in turn impacts

floral community composition (Rosenberg et al. 1983). Typically, slope gradients decreased in relationship to priority species occurrence, suggesting that steep slopes provide structure that is less favorable to this group as a whole. A positive association was also found to exist between focal species and diameter at breast height. Although trees of historical size are not common along the ridge-top habitats at the Jocassee Gorges, focal species occurrence is shown to coincide with larger trees present on the property.

Eastern Wood-Pewee

The eastern wood-pewee was found to be most prolific in sites treated with prescribed fire. When occurrence is compared to that of the control sites, a drastic reduction in occupancy can be observed. Wilson et al. (1995) and Artman et al. (2001) found that this tyrannid increased in response to post burn conditions, possibly due to higher levels of flying arthropods and greater visibility in the midstory. This species breeds in virtually every type of wooded community in the East including both deciduous and coniferous forests. However, there is evidence that use of coniferous habitat may be more common in the South (Peck and James 1987, Post and Gauthreaux 1989, Stevenson and Anderson 1994). Eastern wood-pewees are commonly associated with forest clearings and edges, but tend to avoid streams in Eastern forests (Murray and Stauffer 1995). A study in Wisconsin suggests trends toward higher breeding densities at drier sites, supporting the results of related research (Bond 1957, Robbins et al. 1989).

Inquiry into habitat associations for the eastern wood-pewee at the Jocassee gorges revealed that occurrence was related to percent pine and slope gradient, with a marginally insignificant relation to shrub cover. The positive association with percentage of pine occupancy

within a stand coincides with other studies on avian habitat preference. In the Piedmont of Georgia, Johnston and Odum (1956) discovered that the highest densities of the species resided in pine forests of intermediate age (around 60 years) with lower densities in older pine and mature oak-hickory forests. Ridge-top stands at the Gorges most frequently fall into an intermediate age class, further supporting these findings.

Existing literature on slope gradient preference for the eastern wood-pewee is scarce. For this study, pewees were more commonly detected on moderate slopes of southern aspects where drier conditions are prevalent. There was also a marginally insignificant negative correlation of occurrence with shrub cover which may potentially hinder foraging opportunities for tyrant flycatchers that require more open habitats for catching flying insects on the wing. In the Appalachian mountains of Virginia, eastern wood-pewees were most abundant in forested stands of intermediate age with little understory vegetation, whereas abundance habitually decreased with the presence of tall shrubs. In this case, results from the generalized linear mixed model were commensurate with previous findings connecting eastern wood pewees with more open, pine dominated sites of drier aspect.

Ovenbird

The ovenbird can be a troublesome species to define in terms of habitat association. Conclusions on habitat preference are widely varied. Hamel (1992) suggest that this bird favors deciduous or mixed forests with a moderately dense understory in drier upland areas on moderate slopes, while Kendeigh and Fawver (1981) observed ovenbirds in all forest types in the mountains of North Carolina and Tennessee, but found them to be most abundant in the oak and

pin-oak forests of upper slopes and ridges. Similar results were also observed in western North Carolina, where the ovenbird was found to be most abundant in xeric oak and oak-hickory forest types (Katz 1997). Ovenbirds have also been found to be associated with more exposed slopes on northerly aspects with large trees (James 1971, Smith and Shugart 1987). Structural vegetation of ovenbird territories has been intensively studied. Canopy heights of 16 – 22 m are repeatedly cited as important parameters in habitat selection (Smith 1977, Collins 1983, Thompson and Capen 1988). Data from Jocassee returns similar results. Ovenbirds were found to be positively associated with canopy heights between 18 and 28 m, and positively associated with trees of larger diameter (> 9 cm dbh). These claims are supported by Smith and Shugart (1987), who report similar results.

Percent ground cover is also commonly found to be correlated with ovenbird abundance. Unlike canopy height and diameter at breast height, however, the mean values of these parameters vary widely among studies (Sweeny and Dijak 1985, Thompson and Capen 1988). Most studies involving ovenbirds and fire have reported declines in response to prescribed fire or mechanical treatments (Wilson et al. 1995, Rodewald and Smith 1998, Artman et al. 2001). Few studies have, however, reported no response to burning. Those that have not reported response to burning also show positive correlations to leaf litter depth and live tree density, two factors often affected by fire (Greenberg et al. 2007). Occurrence data from Jocassee suggests that ovenbirds were not affected when comparing burned and control sites for the first year of sampling, and were potentially benefited by the second year of sampling as they were present in larger number in treatment areas. Low severity burns may be an explanation for this observed trend. As litter depth and other related terrain data were not collected, it is difficult to interpret this species' movements in the area when compared to past research. A likely

explanation for the increase in burn plot occupancy is an increase in availability of food resources and retention of cover following low severity burns.

Worm-Eating Warbler

Occurrence of the worm-eating warbler often coincides with large tracts of mature deciduous or mixed deciduous-coniferous forests overlapping hillsides with patchy shrub layers (Keller and Yahner 2007). Previous research suggests that plant composition of the forest community may be less important to this species than forest age/size, presence of topographical variation, and presence of dense patches of shrub cover (Hall 1983, Gale et al. 1997). Individuals of this species occur in a plethora of forest communities, and may be found in a variety of mesic to xeric environments.

Prescribed fire has the potential to remove or greatly reduce understory vegetation, leaf litter, and duff accumulation. This consequence often leads to short-term declines of ground/shrub nesting species like the worm-eating warbler. Studies in Missouri (Blake 2004) and Ohio (Artman et al. 2001) report a near absence of worm-eating warblers in treatment plots after treatment as fire often alters the structure and composition of the understory in such a way that reduces nesting and foraging opportunities for this species. Greenberg et al. (2007) proposed that leaf litter removal associated with burns was a catalyst to the reduction in numbers of breeding worm-eating warblers in North Carolina, as they found that prescribed burns likely caused a decrease in the density of this warbler.

At Jocassee, worm-eating warbler occurrence was found to decrease immediately after fire. For the first year of sampling, where treatment plots were burned a maximum of one year

prior to surveying, this species was more commonly detected in areas that were not impacted by fire. In the second year of sampling, where many of the sampling points were located in plots that had not experienced fire within two years prior, worm-eating warbler numbers in the burned treatment areas increased, becoming more comparable to control areas in terms of the density of this species.

Of the variables included in the regression analysis, only shrub cover and percent pine were found to be important in predicting the occurrence of this species at Jocassee. Slope gradient, although not concluded to be of statistical significance within the regression model, is a notable variable in that increasing values correlate positively with worm-eating warbler occurrence and only remained outside of the model by a small margin. Reductions in understory tend to affect worm-eating warblers negatively (Rodewald and Smith 1998), as the species often relies on the presence of patchy or thick shrub layers for nesting and foraging.

Worm-eating warblers responded negatively to increases in pine occupancy within a stand for this study. Supporting research has suggested that worm-eating warblers occur more infrequently in pine forests than in hardwood dominated forests (James and Neal 1986, McNair and Post 1993). Typically, pine stands are associated with reductions in vertical structure and canopy cover, two important variables of which higher values are positively related to occurrence as reported from past studies (Wenny et al. 1993, Gale et al. 1997). In regards to landform, moderate to steep slope gradients are a common characteristic of worm-eating warbler habitat throughout the breeding range of this species (Mengel 1965, Hall 1983, James and Neal 1986, Wilcove and Robinson 1990, Wenny et al. 1993, Gale et al. 1997). Descriptions of nest site selection from flat coastal sites also suggest that worm-eating warblers build nests on any available slope, regardless of size (Hess et al. 1999). Existing literature supports the findings of

this study concluding that worm eating warbler occurrence is likely related to the habitat variables included in the model.

Scarlet Tanager

Although scarlet tanagers were not found to have any significant relationships to habitat characteristics at Jocassee, a loose association with canopy height is worth mentioning. The inconclusiveness in examination of habitat preference for this tanager species is similar to that of Katz (1997), who found that scarlet tanagers in North Carolina did not show preferences for any particular habitat characteristic using a landscape ecosystem classification model. This species is most commonly associated with mature deciduous forests in uplands, especially where oaks and large trees are common as well as in moist, mixed mesophytic forests of slopes and ravines in the mountains in the southern portion of their range (Hamel 1992, Mowbray 1999). At Jocassee, scarlet tanager populations were virtually homogeneous across the landscape of the upper slope positions regardless of presence in treatment or control sites. A slight increase in number of individuals within the treatment areas, although not statistically relevant, is apparent when observing raw occurrence data for each sampling year.

Rush et al. (2011) discovered that the distribution of scarlet tanagers did not vary considerably in response to fire severity or time since fire. Studies have, however, found that this species is related to many habitat variables affected by fire. In Arkansas, James (1971) concluded that the occurrence of scarlet tanagers is correlated with number of tree species, percentage of canopy cover, canopy height, number of medium sized trees and/or decreasing shrub density, and the presence of large isolated trees. Shy (1984) also found that scarlet

tanagers had a tendency to associate with stands of higher canopy height and number of tree species, further supporting the loose association with increased canopy height detected in this study. It is unclear why habitat associations for this species were not detected at Jocassee. One possibility is that there was not enough structural variation in the limited number of vegetation plots to discern affinity for particular habitat characteristics. A high rate of detection for scarlet tanagers in both the burned and control areas may also be a reason that specific associations were not found.

Hooded Warbler

Hooded warblers typically inhabit mature forests and moist ravines with prevalent shrub layers and trees large enough to create significant tree gaps (Hamel 1992, Evans Ogden and Stuchbury 1994). In North Carolina and Tennessee, this species was found to be most abundant in oak-dominated forest types, but did occur in lower numbers in pine-oak and hemlock forests (Kendeigh and Fawyer 1981, Wilcove 1988). At Jocassee, hooded warblers were found in both burned and unburned habitats. Data comparing the two site types shows a significantly negative response to the effects of fire. As this species is most commonly associated with sites in which fire is not common, it is possible to assume that fire has little potential benefit in the short term.

The response of hooded warblers to fire has been well documented within the first few years after fire. Most of the literature existing on the subject reports that this species is adversely affected by fire (Greenberg et al. 2007). Artman et al. (2001) reported that hooded warblers declined in response to fire and did not recover within one year after burning. However,

prescribed burning in an Indiana forest did not appear to have negative effects (Aquilani et al. 2000). The contradictory reports on the response of this species to burning may be explained by the level of fire severity from which conclusions were drawn. Rush et al. (2011) found that the density of hooded warbler was highest in low severity burns and in plots with greater time since fire. Conversely, plots exposed to high severity fires supported much lower numbers of hooded warblers as did those plots where fire had recent impacts.

Hooded warblers were also found to be positively correlated with shrub cover at Jocassee as suggested by Greenberg et al. (2007). Territories of this species typically include small clearings where a suitable shrub understory is available for nesting. In many cases, local populations have declined dramatically as shrub layer was reduced (Bent 1953, Gartshore 1988, Bisson and Stutchbury 1999, Oberholser 1972, Eaton 1988, Sibley 1993). Fire often reduces shrub layer immediately after the disturbance event, supporting a shift in hooded warbler populations toward areas with no recent fire history. The model most suitable for predicting hooded warbler occurrence for the study site also includes diameter at breast height and crown diameter. As a mature forest species, hooded warblers are often associated with larger trees with larger crown diameters, evidence supporting the findings of this study. Although the burned and control areas were similar in terms of average diameter at breast height and average crown diameter, reduction in shrub layer stemming from fire are likely the cause of short term decreases in hooded warbler occurrence.

Black and White Warbler

Katz (1997) reported black and white warbler occurrence to be highest in pine-oak/heath forest types in North Carolina. Kendeigh and Fawver (1981) and Wilcove (1988) also found a relation to mixed forest types such as chestnut oak and mature pine-oak in the Great Smokey Mountains where this species has shown recent increases in number. Black and white warblers are often thought to associate mainly with deciduous species. Certain studies have suggested that this may not always be true. Clark et al. (1983) found that this species was often associated with high conifer volumes at the northern end of its range. In Texas, black and white warblers show preference for mature pine-hardwood forests (Conner et al. 1983).

In response to burning, studies have shown that black and white warblers do not commonly alter their distribution. This species, which nests on the ground and forages on tree trunks and branches, has been found to decrease in response to mechanical treatments, but does not appear to be adversely affected by burns in some studies (Greenberg et al. 2007, Artman et al. 2001, Wood et al. 2004). In contrast, Wilson et al. (1995) reported declines in black and white warbler populations in response to prescribed burns in Arkansas, USA.

Black and white warblers at Jocassee were found in like number for both the burned sites and the control sites, similar findings to previously mentioned studies. Two significant variables related to occurrence were identified through regression analysis including number of snags and canopy cover. Crown diameter and canopy height are additional variables possibly related to occurrence, but did not meet the criteria to remain in the model. Conner et al. (1983) also reports that increasing canopy cover, along with large tree density and number of tree species is related to the presence of black and white warblers. Although these characteristics do not show significant variation between burned and control sites, continued burning of sites at Jocassee may reduce numbers of this species if the suggested habitat associations are valid.

Continued fire management on xeric sites will likely result in reduced canopy cover and a decrease in the number of tree species as many cannot withstand frequent fire.

Literature connecting black and white warblers to snag density is scarce. This species is often described as a specialist wood-warbler, focusing the bulk of its foraging on the bark of tree trunks and large limbs. Morse (1989) claims that it is more accurate to define this species as a foraging generalist because it frequently joins other warblers in gleaning outer branches and foliage. Through a study of substrate use, Morse (1989) indicates that black and white warblers spend most of their time foraging along dead limbs, followed by large inner limbs, trunk, tip of vegetation, and small outer parts of limbs. Thus, though it displays specialist tendencies as a bark forager, it still retains other foraging skills more typical of most other warblers. In this study, black and white warblers are most likely associated to snag density as a result of diversified food resources. Dead trees often provide easier access to insects which may not be readily available with live trees. Being a foraging generalist may help account for this species' relatively equal abundance between treatment and control sites.

Black-Throated Green Warbler

The black-throated green warbler was the most commonly detected species of concern at Jocassee across sampling years. In the mountains, this species has a wide range, from boreal coniferous forests in the northern portion of their range to mixed coniferous/deciduous forests and completely deciduous forests in the southern portion (Collins 1983, Morse 1993, Robichaud and Villard 1999). In the Great Smoky Mountains, the black-throated green warbler was found most commonly in hemlock/deciduous forests, but was also present in small numbers in cove

and oak-dominated forests (Kendeigh and Fawver 1981). Hamel (1992) describes the preferred habitats of this warbler in the Southern Appalachians as coniferous and mixed forests, especially those that contain hemlock and white pine.

At Jocassee, fire was shown to have a significant impact on the distribution of black-throated green warblers. Although the preferred habitats of this species may be those of coniferous and mixed forests in the Southern Appalachians, as indicated by Hamel (1992), disturbance events conducive to the proliferation of these habitat types were found to be detrimental to this species for the short term in this study. One possible explanation for this trend is a greater association with white pine species in the region. White pines (*Pinus strobus*) are often negatively impacted by fire, while yellow pine species of xeric ridges tend to benefit from the management practice of controlled burning. A positive correlation to pine occurrence is supported by proposed habitat preference for the Southern Appalachians.

Altitude was also found to be of importance in predicting the occurrence of this species at Jocassee. The Southern Blue Ridge Mountains serve as the extreme lower limit of the black-throated green warbler's range. Populations are restricted to this mountain chain at lower latitudes, suggesting that elevation and topographical influence are limiting factors to the occurrence of this species. Elevation also has implications to species composition of vegetation and food availability, which are inevitably important components of the range of any given species. It is unclear what the long term effects of fire on black-throated green populations are from current data. Populations will likely respond negatively as vertical structure and number of tree species is reduced as a result of frequent fire. Further investigation is necessary to draw conclusions from these observations as the response of this species to disturbance management is not fully understood.

Foraging and Nesting Guilds

For the year of 2012, and for years combined, variation in habitat usage for foliage-gleaners and hawkers were significant in comparisons made between burned treatment sites and control sites. Foliage gleaners, including black-throated green warblers, worm-eating warblers, and hooded warblers, were found to have greater statistical abundances in the control sites at Jocassee. This outcome may be expected, as this group generally requires shrubby or broad leaved vegetation on which to forage. Both hooded warblers and worm-eating warblers feed primarily in shrubs, and often decline in response to fuel reduction treatments (Evans et al. 1994, Gale 1995).

Statistical differences in the mean number of hawking species between burned and control sites are directly related to the occurrence of eastern wood pewees and great crested flycatchers. Eastern wood pewees are known to breed in a variety of habitats, but are associated with open midstories that are beneficial to foraging success of species that require vertical and horizontal space to feed efficiently. Reductions in midstory, leading to greater visibility and higher levels of flying arthropod diversity (Wilson et al. 1995, Artman et al. 2001), are probable explanations for the selection of burned habitats over more dense control sites for the hawking guild.

No differences were detected in bark-gleaners or in ground-foragers between the treatment and control sites. Bark-gleaners were likely not affected by controlled burning at Jocassee as resulting tree mortality was rare, allowing a stand impacted by fire to retain compositions of live trees similar to those of control stands. This similarity in tree composition

denotes a likeness in the composition of prey species, which ultimately controls predator distributions. Black and white warblers and woodpeckers comprise the majority of this guild within the study site, none of which exhibited higher statistical occurrence in either the treatment or the control areas. Ground foragers were also likely not affected by burning as a result of low severity fires which often impact leaf litter, yet tend to leave duff layers intact. As suggested by Woinarski (1990), fires may actually enhance food resource availability for ground foraging birds by exposing insects.

For the year of 2012, and for years combined, variation in habitat usage for canopy nesters was significant in comparisons made between burned and control sites. For this study, characteristics of the canopy, such as height, diameter, and cover, did not statistically differ between treatment and control. Fire did not tend to directly impact the overstory of most stands at Jocassee, leading to the conclusion that the foraging habits of canopy nesters, such as black-throated green warblers, may be dictating what nest sites are selected. Birds often claim nesting territories near food sources, and in this case, it is unlikely that a change in canopy structure is the cause of shifts in the distribution of canopy nesters. An explanation for the stability in cavity nesters and ground/shrub nesters between sites at Jocassee may also be a result of low intensity fires that did not alter habitat structure in a way that members of these guilds were either attracted or displaced by controlled burning.

When considering time since fire, foliage gleaners preferred plots with greater distance, along a timeline, from burn events. As shrubs generally require time to regenerate after fire, foliage gleaners are often displaced immediately following disturbance as a result in reductions in the midstory and the understory, areas in which this guild often forage. Both bark-gleaners and hawkers were found to occur commonly in plots with 0-1 years since fire. Open habitats

created by burning were remnant during sampling of these recently burned plots, while plots burned one year prior were allowed more time to regenerate, closing gaps used for foraging by hawks. Bark-gleaners likely benefited from a new crop of dead limbs on which to forage where fires had impact.

Canopy nesters were detected more often in plots with a greater time since fire, most likely due to a re-emergence in the availability of foraging resources after regeneration. Cavity nesters, however, were found in greater number in plots burned in 2011. This may be a result newly created snags or scars within standing trees. No significant differences were detected for ground/shrub nesters, suggesting that fire did not have a large enough impact on this vegetation layer to alter the distribution of this guild.

Visualization

3D visualization can be successfully used as an aid to understand the processes of nature. Visualization allows users to explore forest stand conditions of the past, present, and future by integrating raw data with virtual environments. Such a tool is not only beneficial to the field of science, but to land managers and to the public. 3D visualization provides a glimpse at the consequences of management, both positive and negative, allowing viewers to comprehend the means and ends of long-term forest management.

The approach used in this study produced still images of control stands under present conditions, stands immediately following fire, stands after 2 years of fire, and stands under long-term fire-management. As the accuracy and quality of visualizations are essential to managers and land owners (Daniel and Meitner, 2001), it was imperative that high quality tree images were

used to create photorealistic images. The image of a control stand at Jocassee depicts a stand of higher basal area with near complete canopy coverage and a moderate shrub layer. This image represents a common stand at the study site that has not been impacted by fire. The visualization of a stand directly after fire is a picture of what most may think of when they consider forest fires. Vegetation is reduced and some trees are left dead or scarred from the flames. This is typically the image on which the public focuses. The visualization, however, serves to show that these consequences are only temporary, and that regeneration is imminent.

The image of a stand 2 years after fire shows that stand undergoing the process of regeneration. In this visualization canopy gaps have been created, reducing basal area and canopy cover and allowing the regeneration of shade-intolerant species along with herbaceous vegetation and shrubs. The final generated image represents a stand that has been long managed by fire. Shade-intolerant species dominate the canopy above a diminished midstory and an understory of grasses and shrubs. The vegetation structure between the visualization of the control areas and the final image is drastically different. The production of multiple images allows viewers to see a gradual change in stand composition, and that fire is not detrimental when used as a management tool.

3D visualization is gaining popularity as an asset to forest management plans and other landscape activities (McCarter 1997, McGauphey 1998, Sheppard et al, 2004). New advancements in technology are increasing the capabilities of this software. LIDAR or light detection and ranging, is a relatively new tool that can rapidly measure the features of the Earth's surface. Technology such as this allows one to obtain large datasets on forest stand information. The combination of LIDAR and visualization software may prove to be very useful in future explorations of forest structure. As more research is conducted on the usefulness of 3D

visualization software, its popularity as an asset to scientists, managers, and to those wishing to raise public awareness will continue to grow.

CHAPTER V

CONCLUSIONS

Results from this study indicate that fire can be a useful tool for manipulating habitat structure and management on the avian community level as well as the individual species level in the mountains of South Carolina. It is difficult to determine how certain species will react to the labored effects of controlled burning in the long term. It is apparent, however, that this form of disturbance serves to diversify structure across the landscape, providing new sources for both nesting and foraging. The effects of fire severity on avian community composition in the region may also prove to be of great importance with further research.

The visualization component of this study has shown to be incredibly relevant to management of this type. Public perception of fires is often focused on the immediate aftereffects where aesthetical appeal is low. Virtual environments created through this software allow managers to convey forecasted stand conditions in years following fire to people of all backgrounds. Support is a large part of disturbance management on lands frequented by the public, and must be achieved to carry out practices that promote biotic diversity and sustainability.

Many factors such as competition, fluctuating food resources, and the evolutionary history of the species, can affect its use of a preferred habitat. This study only scratches the surface of the great intricacies existing within the avian biological community at the Jocassee Gorges. As a result of the short nature of the research, all findings reported should be considered preliminary.

APPENDICES

Table A-1. Mean values of 13 environmental variables measured for treatment and control sites.

Variable	Burn	Control	Plots Burned in 2010	Plots Burned in 2011
Altitude	640.97	574.79	677.14	581.05
Gradient	10.18	10.34	10.19	10.21
Basal area	99.53	127.98	99.61	99.65
Number trees	25.61	31.91	25.96	25.13
Number snags	4.43	2.85	4.84	3.75
DBH	9.85	9.86	9.76	10
Percent pine	31.42	28.99	33.38	28.29
Canopy cover	85.09	94.15	83.86	87.17
Crown diameter	26.44	25.58	25.92	27.33
Canopy height	67.09	70.71	63.44	73.26
Herbaceous cover	7.72	4.37	8.01	7.12
Shrub cover	12.95	11.36	14.73	4.45
Ground cover	23.23	18.84	25.38	5.69

Table B-1. List of all species recorded by 50m radius point counts in each year for treatment and control sites.

Species	Burn 2010	Control 2010	Burn 2011	Control 2011
Acadian flycatcher <i>Empidonax virescens</i>	X	X	X	X
American crow <i>Corvus brachyrhynchos</i>	X	X	X	X
American goldfinch <i>Carduelis tristis</i>		X	X	X
American redstart <i>Setophaga ruticilla</i>	X		X	X
American robin <i>Turdus migratorius</i>			X	
Barn swallow <i>Hirundo rustica</i>	X	X	X	X
Barred owl <i>Strix varia</i>	X			
Black and white warbler <i>Mniotilta varia</i>	X	X	X	X
Black-throated green warbler <i>Dendroica virens</i>	X	X	X	X
Blackpoll warbler <i>Setophaga striata</i>	X			
Blue-gray gnatcatcher <i>Poliaoptila caerulea</i>	X	X	X	X
Blue-headed vireo <i>Vireo solitarius</i>	X	X	X	X
Blue jay <i>Cyanocitta cristata</i>	X	X	X	X
Broad-winged hawk <i>Buteo platypterus</i>	X		X	
Brown-headed nuthatch <i>Sitta pusilla</i>		X		X
Brown thrasher <i>Toxostoma rufum</i>	X		X	X
Cape may warbler <i>Setophaga tigrina</i>	X		X	
Carolina chickadee <i>Parus carolinensis</i>	X	X	X	X
Carolina wren <i>Thryothorus ludovicianus</i>	X	X	X	X
Chimney swift <i>Chaetura pelagica</i>	X	X	X	X
Common raven <i>Corvus corax</i>	X	X	X	X
Common yellowthroat <i>Geothlypis trichas</i>		X		
Downy woodpecker <i>Picoides pubescens</i>	X	X	X	X
Eastern bluebird <i>Sialia sialis</i>			X	
Eastern phoebe <i>Sayornis phoebe</i>	X		X	X
Eastern towhee <i>Pipilo erythrophthalmus</i>	X	X	X	X
Eastern wood-peewee <i>Contopus virens</i>	X	X	X	X
Fish crow <i>Corvus ossifragus</i>	X			
Great-crested flycatcher <i>Myiarchus crinitus</i>	X	X	X	
Great-horned owl <i>Bubo virginianus</i>		X		
Hairy woodpecker <i>Picoides pubescens</i>	X	X	X	X
Hooded warbler <i>Wilsonia citrine</i>	X	X	X	X
Indigo bunting <i>Passerina cyanea</i>	X	X	X	X
Louisiana waterthrush <i>Parkesia motacilla</i>	X		X	

Table B-1. List of all species recorded by 50m radius point counts in each year for treatment and control sites (Continued).

Species	Burn 2010	Control 2010	Burn 2011	Control 2011
Mourning dove <i>Zenaida macroura</i>	X	X	X	X
Northern cardinal <i>Cardinalis cardinalis</i>	X	X	X	X
Northern flicker <i>Colaptes auratus</i>	X	X		
Northern parula <i>Parula americana</i>	X	X		
Ovenbird <i>Seiurus aurocapillus</i>	X	X	X	X
Peregrine falcon <i>Falco peregrinus</i>		X		X
Pileated woodpecker <i>Dryocopus pileatus</i>	X	X	X	X
Pine warbler <i>Setophaga pinus</i>	X	X	X	X
Purple Martin <i>Progne subis</i>				X
Red-bellied woodpecker <i>Melanerpes carolinus</i>	X	X	X	X
Red-breasted nuthatch <i>Sitta canadensis</i>			X	X
Red-eyed vireo <i>Vireo olivaceus</i>	X	X	X	X
Red-shouldered hawk <i>Buteo lineatus</i>	X			
Ruby-throated hummingbird <i>Archilochus colubris</i>	X		X	
Scarlet tanager <i>Piranga olivacea</i>	X	X	X	X
Summer tanager <i>Piranga rubra</i>	X	X		
Swainson's thrush <i>Catharus ustulatus</i>	X			
Tufted titmouse <i>Baeolophus bicolor</i>	X	X	X	X
White-breasted nuthatch <i>Sitta carolinensis</i>	X	X	X	X
White-eyed vireo <i>Vireo griseus</i>		X		
Wild turkey <i>Meleagris gallopavo</i>	X	X	X	X
Wood thrush <i>Hylocichla mustelina</i>			X	
Worm-eating warbler <i>Helmitheros vermivorous</i>	X	X	X	X
Yellow-billed cuckoo <i>Coccyzus americanus</i>	X	X	X	X

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