EFFECTS OF PRESCRIBED BURNING ON BREEDING BIRD POPULATIONS OF THE DANIEL BOONE NATIONAL FOREST PIONEER WEAPONS AREA

A Thesis

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by

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THE EFFECT OF PRESCRIBED BURNS ON BREEDING BIRD POPULATIONS WITHIN THE PIONEER WEAPONS AREA OF THE DANIEL BOONE NATIONAL FOREST.

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The use of prescribed burns is experimentally being used as a tool for forest vegetation management to maintain and restore oak-dominated (*Quercus* spp.) forests in Eastern Kentucky. Prescribed burns were conducted in the spring of 2003 within the Pioneer Weapons Area of the Daniel Boone National Forest. Breeding bird communities of three burned sites and three unburned sites were studied in the summers of 2002 and 2003 to determine the effects of fire on breeding birds. Burned sites had lower average species abundance and average species richness while control sites showed small gains in these indices. Both control and treatment sites showed small gains in Shannon's Diversity Index in 2003. Of the 39 bird species captured, ground nesting, low shrub nesting and mid-story nesting canopy species abundances were all negatively affected in burned sites, while unburned sites showed small increases. Upper canopy nesting and cavity nesting species abundance did not decline in both control and treatment sites. Rock nesting species increased substantially within the burned sites but did not differ within unburned sites. Population densities of Ovenbirds (*Seiurus aurocapillus*) and Hooded Warblers (*Wilsonia citrina*) showed the greatest decline while population densities of Eastern

Phoebes (*Sayornis phoebe*) had the highest increase. Total abundance of the Brown Headed Cowbird (*Molothrus ater*), a brood parasite, increased in burned sites while in unburned sites its abundance did not change. Results suggest that the initial effects of prescribed burning are likely to have negative effects on ground, low shrub and ground nesting species, but initial effects on upper canopy and cavity nesting are likely to be minimal. Continued monitoring would be appropriate to assess further responses to bird communities associated with potential long-term effects of prescribed burning.

Accepted by: David J. Eisenhour, Chair

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Allen C. Risk

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CHAPTER I

INTRODUCTION

In the United States, migratory breeding bird densities are highest within the eastern forests of the Appalachian Mountains (Terborgh, 1992). Throughout the last two decades there has been great concern over the decline of migratory breeding bird densities within the eastern U.S. (Artman et al., 2001). One potential cause of this decline is the near complete transformation the eastern U.S. forest has undergone during the last century (Artman et al., 2001; Rodewald and Adams, 2002). One of the major factors in tree species composition shift is the near absence of periodic forest fires. In the absence of fire, fast-growing, fire-intolerant tree species have invaded and out competed native hardwood species, ultimately changing the dynamics of the forest canopy levels (Aquilani et al., 2000; Elliott et al., 1999). To encourage the regeneration of fire-tolerant species, the Morehead Ranger District in the Daniel Boone National Forest (DBNF) has started conducting prescribed burns within sections of the Pioneer Weapons Area of the DBNF.

Prescribed fire has been shown to effectively alter succession in various communities, although the majority of studies on prescribed burning on ecosystems has concentrated on grasslands and savanna habitats in the western United States. Effects of prescribed fire have not been widely studied in the deciduous hardwood forests of the eastern United States (Franklin et al., 2003). In addition to the concern in the vegetation shift from fire tolerant species to fire-intolerant species are the interconnected processes associated with the oak-hickory ecosystem including wildlife biodiversity, soil productivity, and forest canopy composition (Artman et al., 2001; Barnes and Van-Lear, 1998).

The decline of migratory breeding bird numbers is one of the more troubling issues facing wildlife management (Artman et al., 2001). As a result, a large amount of information has been collected on migrating bird species, especially neo-tropical migrant species. Yet, there is little information dealing with the effects of prescribed burns, especially in deciduous hardwood forest. Studies have found that small, fast burning, surface fires consume mostly leaf litter and brush, while having little if any effect initially on upper story vegetation. Birds, such as ground and low shrub nesting species, that utilize lower story vegetation have been found to be the most affected (Artman et al., 2001).

An objective of this research is to determine the effect of prescribed burns on migratory breeding birds by comparing bird abundances, diversity, and species richness of different feeding and nesting guilds before and after a burn in a deciduous hardwood forest in eastern Kentucky. In addition, these measures were used in comparisons between burned and nonburned areas. This will allow wildlife managers to better understand the initial effects fire can have on the avian fauna of a hardwood forest ecosystem.

CHAPTER 2

LITERATURE REVIEW

2.1 The Importance of Fire in the Eastern Deciduous Forests

Before European settlement, the eastern deciduous forests of the U.S. had been dominated by oak (*Quercus*) and hickory (*Carya*) species for the last 10,000 years (Franklin et al., 2003; Dey and Guyette, 2000). The abundance of oak species was associated with increased temperature, less precipitation, and an increased fire frequency after the last ice age (Abrams, 2003; Lanham et al. 2002). In addition to climatic changes, forest ecosystems of the eastern United States have been shaped by natural disturbances such as windstorms, ice, landslides, insects, disease, and fire (Elliott et al., 1999).

Fire is an important factor in the structure and species composition of many forest types in the United States, as well as the rest of the world. Of the many different types of terrestrial ecosystems relatively few have not been affected by fire (Franklin et al., 2003). For most environments, fire is infrequent but is such a strong ecological force that even at an incidence level as low as once every century will have major impacts on habitat; thus it may be a key determinant of environmental makeup (Franklin et al., 2003; Dey and Guyette, 2000). Fire occurrence varies widely even among local habitats within ecosystems. Dry ridge tops may have a fire frequency of every few decades, but moist riparian habitats and valleys in adjacent areas may burn only once every few centuries (Franklin et al., 2003; Blankenship and Arthur, 1999).

General theories about the biological affects of fire are undependable because of variations in the regularity, severity, strength, and type of fire.

Disturbances have been widely explored as a vital factor in a continuing struggle to comprehend successional change throughout many of the world's forests (Abrams, 2003; Lanham et al., 2002). Environmental conditions had favored slow growing oak and hickory species over fast growing, but fire-intolerant tree species (Abrams, 2003). Beginning with the settlement of Europeans, North American forests experienced a major increase in the degree of anthropogenic disturbances compared with those created by Native Americans. Widespread logging, land clearing, and the introduction of exotic insects and diseases all led to rapid transformations in the composition of tree species and forest structure.

Foresters since the early 1900's have traditionally thought of wildfires as a force to be suppressed at all measures (Franklin et al., 2003). In the early 1940s, state and federal policies were created to aid in the suppression of many natural occurring forest fires (Aquilani et al., 2000). Overprotection from fire can result in disadvantageous modifications in the plant community by allowing fire-intolerant plant species to invade fire influenced ecosystems. In addition overprotection allows dead trees and brush, as well as other materials, to build up, leaving the community at risk to severe damage if a significant fire should come (Abrams, 2003). The resulting lack of fire has caused a vegetation shift in many parts of the eastern United States, changing the composition of the deciduous forest from dominance by oaks and hickories to an increasing percentage of maple (*Acer*) species and other fire-intolerant species (Lanham et al., 2002). Fire-intolerant tree species have lower timber value and often are less valuable for wildlife than fire tolerant species, such as acorn and nut-producing oaks and hickories (Elliott et al., 1999). It is important for forest managers to utilize management techniques that restore and preserve hardwood forests that benefit both wildlife and industry.

2.2 Use of Prescribed Burns in Forest and Wildlife Restoration

The use of fire for forest management is not a recent development. Native Americans have historically used fire to destroy brush and trees in order to improve hunting and travel through forested areas (Abrams, 2003; Barnes and Van-Lear, 1998). Fire has been used by some professional foresters to reduce hazardous fuels since the beginning of the 20th century (Lanham et al., 2002). Recently many of the misunderstandings about the use of prescribed fire for forest management purposes have been replaced by research based information (Elliott et al., 1999; Brose and Van-Lear, 1998). As knowledge about the effects of fire in forests ecosystems has grown, the use of prescribed fire as a management tool has grown (Abrams, 2003; Brose and Van-Lear, 1998).

Today prescribed fire is used for various forest management objectives in addition to agricultural and range management purposes (Chandler et al., 1991; Lanham et al., 2002). The use of prescribed burning is a desirable and economically sound practice in most forests throughout the U.S. (Franklin et al., 2003). Few alternative management techniques have been developed that can compete with fire from the standpoint of success and expense (Elliott et al., 1999). Chemical applications typically cost more than ten times as much per acre (Chandler et al., 1991). Mechanical treatments such as disking, chopping, or raking are at least twenty times more expensive. Each of these alternatives also have connected environmental risks, such as habitat damage and soil erosion.

The use of prescribed fires can have unwanted consequences, as well. When conditions are not ideal, prescribed fire can severely harm the actual resource it was intended to profit (Elliott et al., 1999). Prescribed fire is a complicated technique and takes qualified personnel to implement (Chandler et al., 1991). Appropriate judgment and systematic scheduling are required for each area where prescribed burning is considered. The incomplete evaluation of any feature can cause serious problems should the fire break out of established boundaries. A prescribed fire that does not complete its proposed purpose is a cost of both time and resources. In some cases it may be necessary to re-burn as soon as satisfactory conditions exist (Chandler et al., 1991; Brose and Van-Lear, 1998).

The frequency and type of prescribed burn ultimately varies on the objective goals set forth by resource managers (Chandler et al., 1991; Lanham et al., 2002). Reasons for prescribed fire in forest management ultimately depend on the goals to be accomplished (Chandler et al., 1991). Prescribed fire is the most realistic method to reduce dangerous buildup of combustible fuels in forests (Elliott et al., 1999). Wildfires that burn into areas where fuels have been reduced by prescribed burning produce a lesser amount of harm and are much easier to direct (Franklin et al., 2003; Brose and Van-Lear, 1998). The proper time between prescribed burns for fuel reduction varies with several factors, including the degree of fuel buildup, past wildfire history, and the risk of a fire (Chandler et al., 1991; Phillips et al., 2000). The time intervals between prescribed burns can be as often as every year or long as 10-15 years, depending on the habitat and present conditions (Elliott et al., 1999; Brose and Van-Lear, 1998).

Prescribed burns frequently also are used for wildlife habitat management (Aquilani et al., 2000). Periodic burns tend to benefit understory species that require a more open habitat (Rodewald and Smith, 1998). A mosaic of burned and unburned areas tends to maximize edge effect, which promotes a wide and diverse wildlife population (Aquilani et al., 2000). Deer, dove, quail, and turkey are game species that profit from prescribed burns (Ford et al., 1999). Selecting the right size, frequency, and timing of burns which correspond to biological requirements (such as nesting times and vegetative conditions) are important to the successful use of burns to improve wildlife habitat (Aquilani et al., 2000; Artman et al., 2001; Elliott el al., 1999).

Prescribed burning, however, does not always help plant and animal species because artificial fires do not necessarily mimic natural fires (Elliott et al., 1999). The interval between prescribed fires as well as fire intensity itself may differ from those natural fires of the past (Franklin et al., 2003; Phillips et al. 2000). Individual requirements of a species must therefore be understood before a fire can be prescribed to benefit that species (Abrams, 2003).

2.3 Concerns of Burns on Neotropical Migrant Bird Numbers

Some researchers have been particularly concerned about the effects of prescribed burns might have on Neotropical migrant songbirds that breed in U.S. forests (Rodewald and Smith, 1998; Rodewald and Abrams, 2002). Neotropical migrants are birds of the Western Hemisphere that migrate long distances from wintering grounds in the New World Tropics to breeding grounds in North America. The Neotropics are generally defined as the tropical regions of Mexico and Central and South America that lie south of the Tropic of Cancer (Hagan and Johnston, 1989). The 361 species of Neotropical migrant birds include herons, raptors, swallows, warblers, and others (Hagan and Johnston, 1989). Over the past 20 to 30 years, biologists have documented the disturbing declines of many Neotropical migrant bird populations throughout the U.S. (Villard et al., 1995).

Research in North American breeding areas has identified key factors that could possibly make Neotropical migrants susceptible to population declines (Hagan and Johnston, 1989). Regional population declines in vast number of Neotropical migrant species have been attributed to loss and fragmentation of breeding, wintering, and migratory stopover habitats (Hagan and Johnston, 1989; Martin and Finch, 1995; Trzcinski et al. 1999). Neotropical migrant birds are difficult to count accurately due to their large breeding distributions and consequently little historical information exists on their large-scale populations changes (Hagan and Johnston, 1989).

In the U.S., loss and fragmentation of breeding habitat is the key focus of avian researchers. Many Neotropical migrants are habitat specialists that require particularly large tracts of specific habitat types for breeding (Martin and Finch, 1995; Rodewald and Abrams, 2002; Girard et al., 2004). As habitats are lost and fragmented, less suitable habitat becomes available, and threats to adult birds and young increase (Trzcinski et al., 1999). Prescribed burns are a potential way for resource managers to restore and maintain Neotropical breeding habitat types that have been changed or lost (Abrams, 2003).

Most of the research that has examined the effects of prescribed burns on birds has taken place in western and southern portions of the U.S. where fire intervals are much shorter than in other regions of the country. Studies in western states have found that burns affect bird species differently depending on the guild and habitat type studied. Prescribed burns in the western U.S. are a useful tool in eliminating unwanted brush, such as sagebrush (*Artemisia* spp.), from public grasslands. Lekking species such as the Sage Grouse (*Centrocercus urophasianus*) have been found to be negatively affected by prescribed fire, due to declines in male lek attendance (Connelly et al., 2000). Other grassland species such as sparrow, blackbirds, and wrens were found to be unaffected as long as burn intervals were kept above two years (Hul et al., 1997).

Southern studies with prescribed burns have been applied to restore native ecosystems such as prairies and longleaf pine (*Pinus palustris*) forests. Most bird species in these habitat types have benefited from the prescribed burns as compared to mechanical hardwood reduction methods (Provencher et al., 2002). Bachman's Sparrow (*Aimophila aestivalis*) reproduction increased following burns in the Kissimmee River watershed, mainly due to increases in prey items such as grasshoppers species and improved nesting habitat (Shriver and Vickery, 2001). Red-Cockaded Woodpeckers (*Picoides borealis*) increased following burns aimed at restoring longleaf pines again due to improvement in foraging and nesting habitat.

Few studies have focused on deciduous forest habitats in the eastern part of the U.S. where fire intervals are much longer. One study (Aquilani et al., 2000) investigated the effects of prescribed burns on forest birds in four study sites: two in the Wayne National Forest and two in the Vinton Furnace Experimental Forest in south central Ohio. Within the study area 50-75 acre areas were burned frequently (once a year for four years) or infrequently (once every four years).

The study monitored 30 bird species in both burned and unburned areas. The study found that after four years of repeated burning, three ground nesting bird species, Ovenbirds, Worm-Eating Warblers, and Hooded Warblers, declined by more than 80 percent. The study attributed these declines to the decrease of leaf litter, shrubs and saplings required by these birds. Ovenbirds nest on the ground and use leaf litter to build and conceal their nests. Worm-Eating Warblers nest on the ground in high moisture areas, and the drying effects of burns may have made them less suitable for nest sites. Hooded Warblers nest within a few feet of the ground in dense shrub thickets, and the burns eliminated most of these thickets. The study also found that two bird species increased: American Robins and Eastern Wood Pewees. The researchers observed these birds feeding in burned areas, suggesting that fire improved their foraging habitat. The study concluded that long term repeated or large scale prescribed burning could change the songbird community in some eastern hardwood forests.

A similar study in a mature hardwood forest in southern Indiana (Aquilani et al., 2000) found similar results with ground and low shrub nesting species becoming less abundant in areas that were burned compared to adjacent unburned areas. Both studies indicate that planning and research at a landscape level will be needed to attain goals for both vegetative restoration as well as maintenance and improvement of bird species diversity.

2.4 Parasitism due to Prescribed Burns

In North America the decline of many songbird species has been attributed in part to the breeding behavior of the Brown-headed Cowbird (Martin and Finch, 1995). This cowbird is an obligate brood parasite, with the female laying eggs in the nest of other species, and leaving the responsibility of rearing and caring of cowbird young up to the host species (Sibley, 2000). There are five species of parasitic cowbirds; two species are found only in the Neotropics, while the remaining three are found in North America (Sibley, 2000).

Of the three North American species, only the Brown-headed Cowbird is widespread and a serious threat to migrant bird species within the U.S (Martin and Finch, 1995). The breeding strategies of the Brown-headed Cowbird have had a significant impact on many host species (Martin and Finch, 1995). These effects are due in part to the high fecundity of the parasite and the reproductive losses incurred by the host (Sibley, 2000). As a consequence of brood parasitism, many hosts suffer reduced nesting success and/or decreased seasonal fecundity (Martin and Finch, 1995). Brown-headed Cowbirds are open habitat species and require open flying lanes in order to have access to the nest of other birds (Martin and Finch, 1995). Other open habitat species, such as Yellow Warblers, have evolved behaviors which have enabled them to reduce parasitism by cowbirds (Aquilani et al., 2000). Dense forest species have not evolved with cowbirds and do not have behaviors or strategies to cope with the effects of these parasites. The fragmentation of forest by humans has enabled cowbirds to gain access to dense forest species nests, having a significant affect on productivity of the nest of such species. Prescribed burns could possibly open forest habitats that were once too dense for cowbird penetration, leaving dense forest bird species open to parasitism. An accurate assessment of the impacts of cowbird parasitism on individual host species is important for understanding the causes of host population declines (Hagan and Johnston, 1989).

The focus of this study was to look at the initial effects of prescribed burns on forest bird communities in a hardwood deciduous forest. Mean abundances, species richness, and Shannon's diversity index were used to determine differences initially after prescribed burns and comparing burned sites to control sites. Differences in these variables helped determine the effect on the forest bird community as a whole. Individual nesting and foraging guilds of birds were investigated in order to determine if fire had effects restricted to specific ecological groups.

CHAPTER III

MATERIALS AND METHODS

3.1 Study Site

The study was conducted in the Pioneer Weapons Area of the Morehead Ranger District in the Daniel Boone National Forest, Bath County, Kentucky (Figure 1). This area is located on the Upper Cumberland Plateau in Northeastern Kentucky (Woods et al., 2002). The study areas largely consist of mixed deciduous forest with artificially constructed grassy wildlife openings and vernal wildlife ponds. The forest canopy is dominated by oak, hickory, and tulip poplar that have been fragmented into different age groups from logging. The understory is diverse with the major species being red maple (*Acer rubrum*), sassafras (*Sassafras albidum*), pawpaw (*Asimina triloba*), and spicebush (*Lindera benozoin*).

The Pioneer Weapons Area has been divided into three main sections: Chestnut Cliff Area (CC), Wolfe Pen Area (WP), and the Buck Creek (BC) Area (Figure 2). Each of these three sections are subdivided into three subsections consisting of a control subsection (CC), less frequent burn subsection (LF) and frequent burn subsection (F) (Figure 2). These experimental sections were established by the National Forest Service as part of a 7.8 km^2 vegetation study (T. Biebighauser, pers. comm., 2005). Frequent burn subsections are burned every spring, while less frequent burn subsections are scheduled to be burned every three years. For this study two bird banding sites were selected within each of the three main sections section (CCLF, CCCC, BCLF, BCCC, WPF and WPC) thus, for

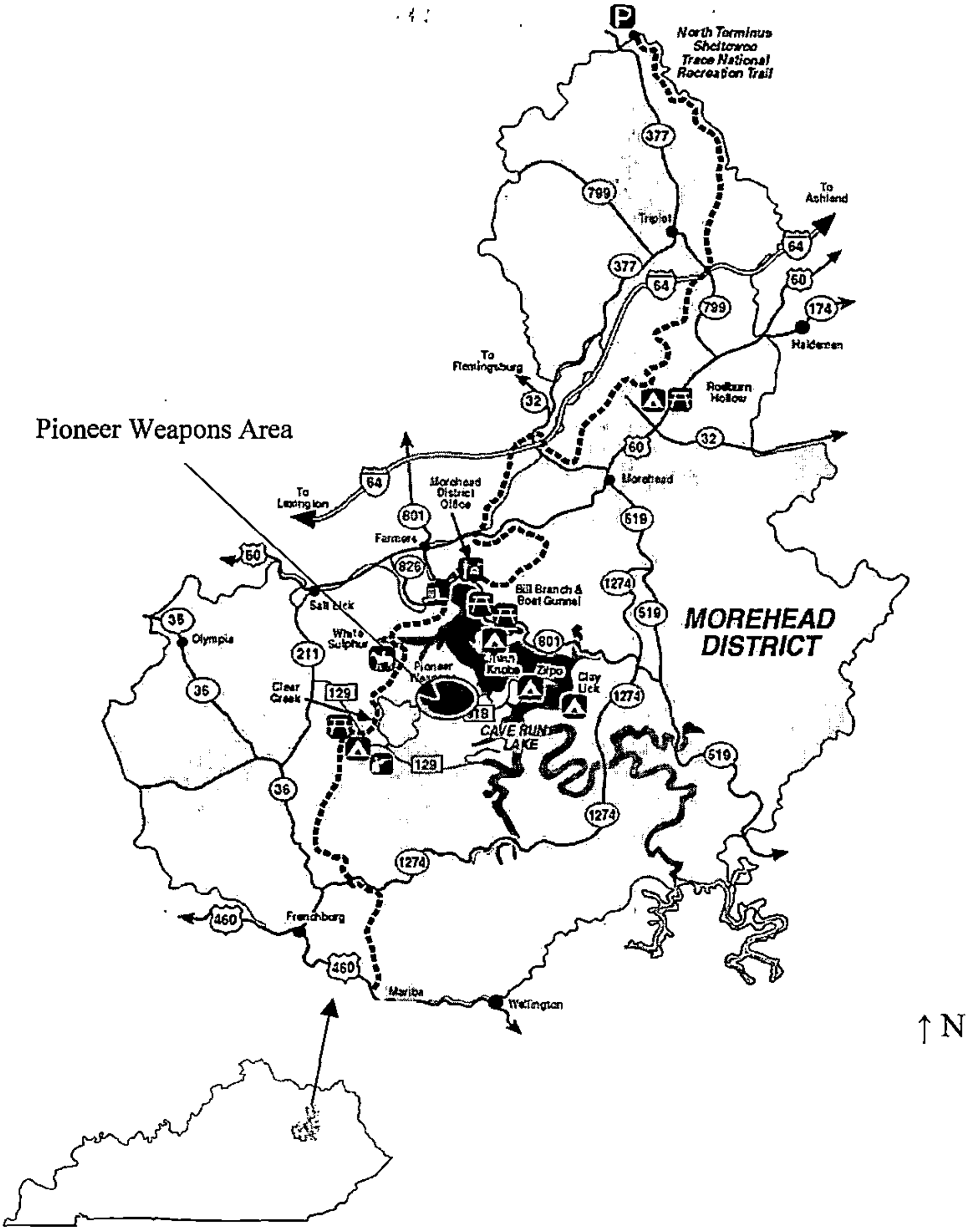

Figure 1. Location of the Pioneer Weapons Area of the Morehead Ranger District of the Daniel Boone National Forest (USDA Forest Service, 2005).

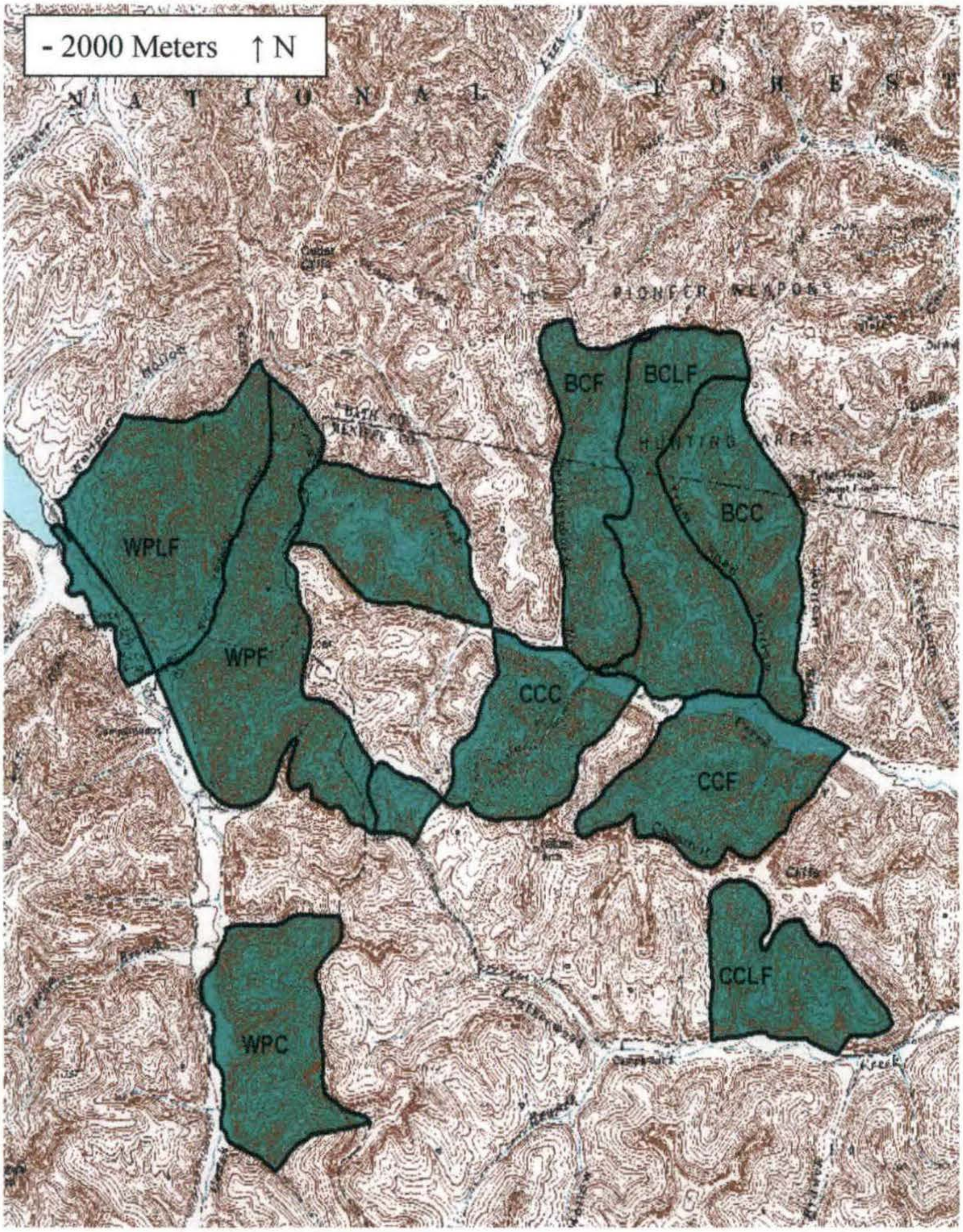

Figure 2. Location of the three experimental sections and subsections within the Pioneer Weapons Area (USGS, 2004).

each of the three main sections one site was located in a burned subsection and one site was located in a control subsection in a location with similar vegetation, elevation, and hydric characteristics (Figure 3). In order to match the environmental characteristics of the treatment sites, two control sites had to be positioned outside the control area boundaries established by the National Forest Service. All bird banding sites were positioned within more mature portions of the forest, away from any recent logging activity.

3.1.1 Buck Creek Section

The Buck Creek Treatment bird banding site was located in the Buck Creek Less Frequent Burn section just off Forestry Service Road 918A (Appendix I). The Buck Creek Control site was located 250 meter north of the northern portion of the Buck Creek Frequent Burn section, on Forestry Service Road 1225 (Appendix I). Both sites are located on ridge tops with grassy wildlife openings and wildlife ponds. Vegetation for both was dominated by an oak and hickory canopy and a sassafras and maple understory.

3.1.2 Chestnut Cliff Section

The Chestnut Cliff Treatment bird banding site was located in the Chestnut Cliff Less Frequent Burn section on Forestry Service Trail 1051 (Appendix I). The Chestnut Cliff Control site was located inside the Chestnut Cliff Control section on Forestry Service Road 918A (Appendix I). Both sites are located on mid-slope portions of hills and contained no grassy wildlife openings or wildlife ponds.

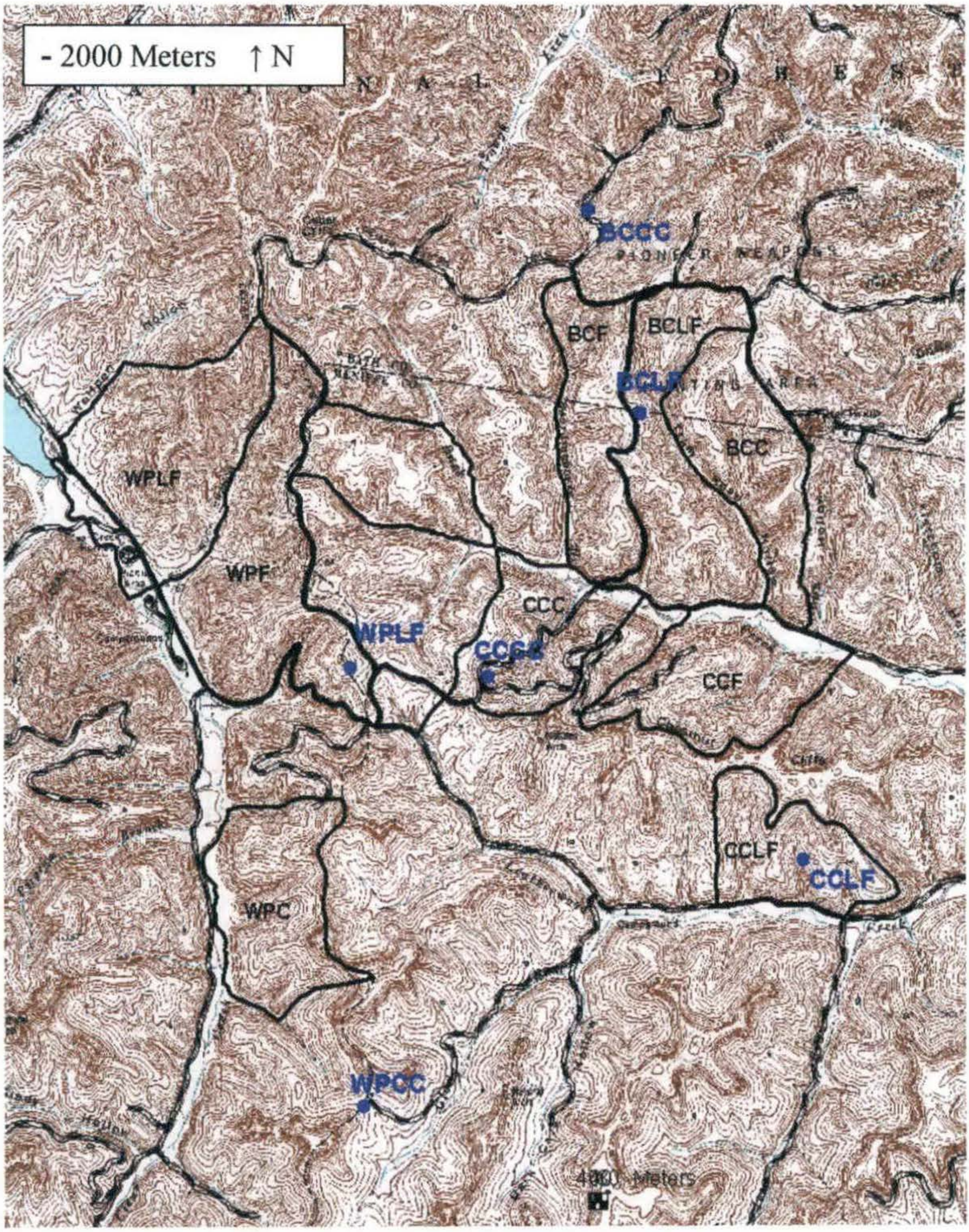

Figure 3. Locations of bird banding sites (USGS, 2004).

Chestnut Cliff control banding site did contain an old logging road. Vegetation for both largely consisted of an oak and hickory canopy with a sassafras and maple understory.

3.1.32 Wolfe Pen Section

The Wolfe Pen Treatment bird banding site was located in the Wolfe Pen Frequent Burn Section located off the Zilpo Scenic Byway entrance (Appendix I). The Wolfe Pen Control site was located South of Wolfe Pen Control section on Forestry Service Road 914 (Appendix I). Both sites were located on mid-slope portions of hills and in riparian areas. The Wolfe Pen Treatment banding site contained no grassy wildlife openings or wildlife ponds, while the Wolfe Pen Control banding site did contain a grassy wildlife opening and wildlife pond. Vegetation for both largely consisted of an oak and hickory canopy with a spicebush and maple understory.

3.2 Field Methods

3.2.1 Mist Net Setup

Birds were surveyed during the breeding season using the Monitoring Avian Productivity and Survivorship (MAPS) bird banding protocol (DeSante et al., 2002). Once a banding site was established mist net sites were chosen along a stretch of forest accessible by foot. Each of the six bird banding sites was sampled with 10 mist nets spread over approximately 100 m. To most efficiently sample bird populations, mist nets were placed opportunistically throughout all sites, including in the vicinity of brush and shrubs, forest edges, water, and grassy openings. To optimize the number of birds and different species captured mist nets were placed relatively uniformly over the available habitat.

Nets used were 12 m x 3 m, 36-mm mesh, four-tier, black, tethered, nylon mist nets (Figure 4). Metal rebar was used to anchor the nets, which were set upright with two three meter sections of metal tubing. Nets were left up for six hours following sunrise; metal tubing and rebar were left anchored in the ground at all times to fix net location.

3.2.2 Bird Banding Methods

The MAPS bird banding protocol divides the North American breeding season into ten banding periods extending from May to August. For this study, bird banding began with Period Three (starting May 21) and ended in Period Ten (starting August 8) in 2002 and 2003. Sites were monitored every banding period throughout the breeding season. Sites were monitored each banding period in no specific order. All sites were sampled beginning at sunrise and ending six hours after that time. In the event of rain or lightning bird banding was delayed or stopped. In addition, individual mist nets that were in direct sunlight were taken down to prevent overheating of captured birds in the net. Mist nets were checked every 15-20 minutes and captured birds were placed in cotton bags and brought to a banding station centrally located at the banding site. All birds captured were banded (if possible) and processed in accordance to MAPS protocol. Birds were identified and processed using Pyle (1997) and Sibley (2000).

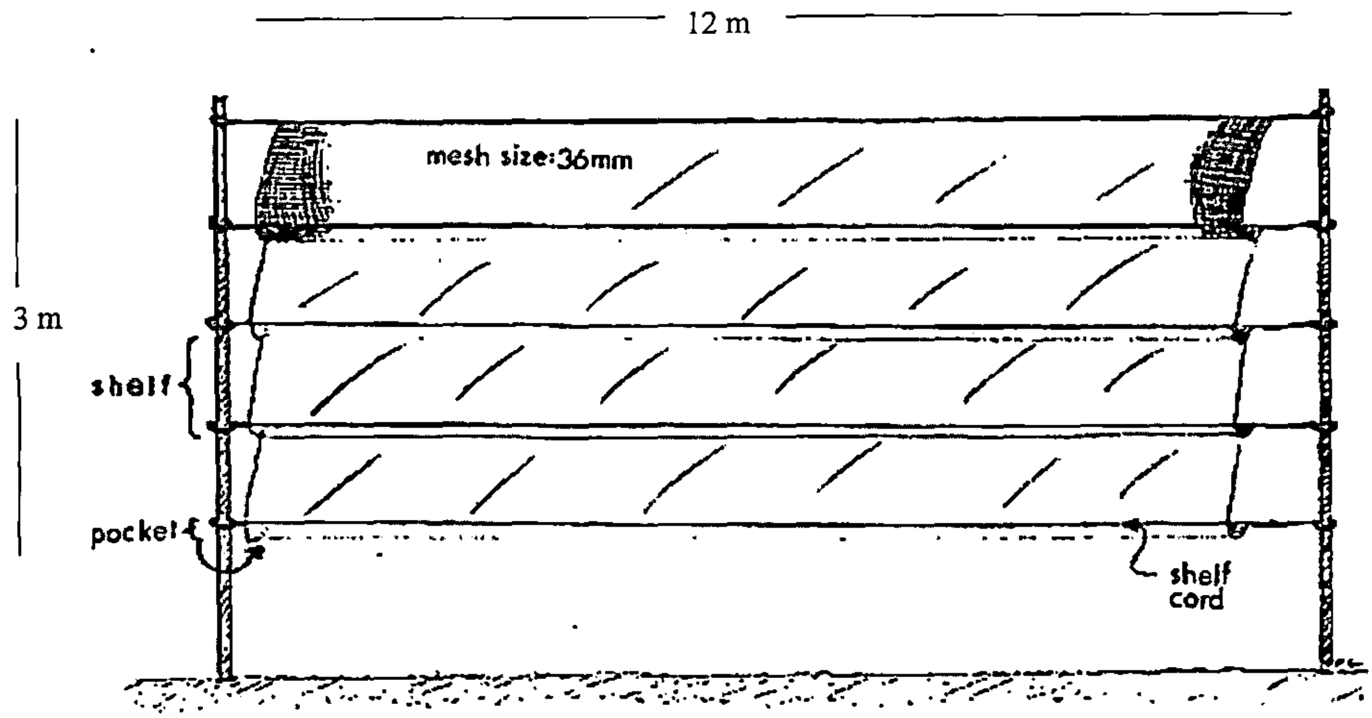

Figure 4. Diagram of a mist net used in the study.

3.2.3 Nesting Guilds

Species captured were classified into six different nesting guilds according to where each species nested (Table 1) (Palmer-Ball, 1996). Categories of nesting guilds included ground nesting, low shrub nesting, midstory nesting, canopy nesting, cavity nesting and rock nesting. Total abundances for nesting guilds were compared for 2002 versus 2003 banding seasons and for treatment versus control sites for 2002 and 2003.

3.2.4 Foraging Guilds

Species captured were classified into six groups according to where each species forages (Table 2) (Palmer-Ball, 1996). Groups include ground foraging, low shrub foraging, midstory foraging, canopy foraging, bole (bark) foraging and aerial (taken in flight) foraging. Mean total abundances for each group were compared for 2002 versus 2003 banding seasons. Mean total abundances for treatment sites versus control sites were also compared for 2002 and 2003 for each foraging guild.

3.2.5 Habitat Assessment

Habitat assessment data was collected at the end of the each banding season (August) using the Monitoring Avian Productivity and Survivorship (MAPS) bird banding protocol for vegetation assessment (DeSante et al., 2002). For each sample site percent cover for all vegetation layers (upperstory, midstory, understory, and ground cover) was determined. Percent cover for each vegetation layer was based on

Table 1. Nesting guilds of species captured throughout the 2002 and 2003 banding seasons (Palmer-Ball, 1996).

Ground Nesting Species

Black-and-white Warbler *(Mniotilta varia)*
Carolina Wren *(Thryothorus ludovicianus)*
Kentucky Warbler *(Oporornis formosus)*
Louisiana Waterthrush *(Seiurus motacilla)*
Ovenbird *(Seiurus aurocapillus)*
Worm-eating Warbler *(Helmitheros vermivora)*

Low Shrub Nesting

Hooded Warbler *(Wilsonia citrina)*
Indigo Bunting *(Passerina cyanea)*
Field Sparrow *(Spizella pusilla)*
Chipping Sparrow *(Spizella passerina)*

Midstory Nesting

Acadian Flycatcher *(Empidonax virescens)*
American Redstart *(Setophaga ruticilla)*
Northern Cardinal *(Cardinalis cardinalis)*
Red-eyed Vireo *(Vireo olivaceus)*
White-eyed Vireo *(Vireo griseus)*
Yellow-throated Vireo *(Vireo flavifrons)*
Wood Thrush *(Hylocichla mustelina)*
Swainson's Thrush *(Catharus ustulatus)*
American Goldfinch *(Carduelis tristis)*

Canopy nesting

Blue-gray Gnatcatcher *(Polioptila caerulea)*
Cerulean Warbler *(Dendroica cerulea)*
Eastern Wood-Pewee *(Contopus virens)*
Scarlet Tanager *(Piranga olivacea)*
Summer Tanager *(Piranga rubra)*
Common Yellowthroat *(Geothlypis trichas)*
Yellow-bellied Flycatcher *(Empidonax flaviventris)*
Magnolia Warbler *(Dendroica magnolia)*
Yellow Warbler *(Dendroica petechia)*

Cavity Nesting

Carolina Chickadee *(Poecile carolinensis)*
Downy Woodpecker *(Picoides pubescens)*
Eastern Bluebird *(Sialia sialis)*
Eastern Tufted Titmouse *(Baeolophus bicolor)*
Hairy Woodpecker *(Picoides villosus)*
Red-bellied Woodpecker *(Melanerpes carolinus)*
White-breasted Nuthatch *(Sitta carolinensis)*

Rock Nesting

Eastern Phoebe *(Sayornis phoebe)*

Table 2. Foraging guilds of species captured throughout the 2002 and 2003 banding seasons (Palmer-Ball, 1996).

Ground Foraging

Carolina Wren *(Thryothorus ludovicianus)*
Louisiana Waterthrush *(Seiurus motacilla)*
Ovenbird *(Seiurus aurocapillus)*
Swainson's Thrush *(Catharus ustulatus)*
Wood Thrush *(Hylocichla mustelina)*

Low Shrub Foraging

American Goldfinch *(Carduelis tristis)*
Chipping Sparrow *(Spizella passerina)*
Field Sparrow *(Spizella pusilla)*
Hooded Warbler *(Wilsonia citrina)*
Indigo Bunting *(Passerina cyanea)*
Kentucky Warbler *(Oporornis formosus)*
Worm-eating Warbler *(Helmitheros vermivora)*

Midstory Foraging

American Redstart *(Setophaga ruticilla)*
Northern Cardinal *(Cardinalis cardinalis)*
Red-eyed Vireo *(Vireo olivaceus)*
White-eyed Vireo *(Vireo griseus)*
Yellow-throated Vireo *(Vireo flavifrons)*

Canopy Foraging

Blue-gray Gnatcatcher *(Polioptila caerulea)*
Carolina Chickadee *(Poecile carolinensis)*
Cerulean Warbler *(Dendroica cerulea)*
Common Yellowthroat *(Geothlypis trichas)*
Eastern Tufted Titmouse *(Baeolophus bicolor)*
Magnolia Warbler *(Dendroica magnolia)*
Scarlet Tanager *(Piranga olivacea)*
Summer Tanager *(Piranga rubra)*
Yellow Warbler *(Dendroica petechia)*

Bole Foraging

Black-and-white Warbler *(Mniotilta varia)*
Downy Woodpecker *(Picoides pubescens)*
Hairy Woodpecker *(Picoides villosus)*
Red-bellied Woodpecker *(Melanerpes carolinus)*
White-breasted Nuthatch *(Sitta carolinensis)*

Aerial Foraging

Acadian Flycatcher *(Empidonax virescens)*
Eastern Bluebird *(Sialia sialis)*
Eastern Phoebe *(Sayornis phoebe)*
Eastern Wood-Pewee *(Contopus virens)*
Yellow-bellied Flycatcher *(Empidonax flaviventris)*

visual estimation. Additional features such as streams, snags, rocks, and ponds were recorded for each site.

3.2.6 Prescribed Burn

Burning took place in March and April 2003, with BCLF burned on April 14, CCLF burned on March 25, and WPLF burned on April 16. Approximately 3.2 km of control lines were constructed by hand prevent to the fire from reaching areas that were not to be burned. Areas where control lines were not constructed used natural features and roads to contain the fire. Burning was accomplished by a helicopter equipped with a plastic sphere dispenser and hand ignition.

3.2.7 Statistical Analysis

Only birds that could take a size 0A to a size 2 U.S. Fish and Wildlife metal bird band were banded in the study and thus used in statistical analysis. Species richness, total abundance, and Shannon's diversity index were calculated and compared for each banding site and treatment for 2002 and 2003. Shannon's diversity index was used because the sample of species abundance was a random sample of the entire forest community. The data were found to be non-parametric by a normality test, all statistical comparisons was performed using a Kruskal-Wallace test ($\alpha \leq 0.05$). Statistical analyses were performed with Minitab Student 14 (Minitab Inc, 2005).

CHAPTER IV

RESULTS

4.1 Comparison of bird diversity, richness and abundance between treatments and years

A total of 895 individual birds encompassing 37 bird species were banded throughout the study (Table 3). During the 2002 banding season a total of 437 birds (31 different species) were banded (Table 3). Bird banding site WPLF had the lowest total abundance (56) and species diversity (0.7378) while the BCLF site had the total highest abundance (114) and diversity (1.4944) (Table 4). For species richness, CCCC had the lowest number of captured species (14) with BCLF having the highest number (21) during the 2002 season (Table 4).

The 2003 banding season yielded a total of 458 banded birds (counting recaptures from the 2002 season) representing 33 different species (Table 5). For species richness CCCC had the lowest number (13) while BCCC had the highest number of captured species (23) (Table 5). CCCC also had the lowest species diversity (0.9096) and CCLF which had the highest diversity (1.1012) (Table 5). For total abundance, CCLF had the fewest captured individuals (52) and BCCC had the most (109) (Table 5). Though there were no significant differences between treatment sites and control sites for total abundance, species richness, or species diversity for 2002 and 2003, total species abundance for treatment sites in 2003 was greatly lower than control sites (Figure 5). There were no significant differences for

Table 3. Numbers of individuals banded for the 2002 and 2003 banding seasons.

Species	2002	2003	Total
Acadian Flycatcher *(Empidonax virescens)*	4	8	12
American Goldfinch *(Carduelis tristis)*	0	2	2
American Redstart *(Setophaga ruticilla)*	17	16	33
Black-and-white Warbler *(Mniotilta varia)*	9	19	28
Blue-gray Gnatcatcher *(Polioptila caerulea)*	4	7	11
Brown-headed Cowbird *(Molothrus ater)*	2	10	12
Carolina Chickadee *(Poecile carolinensis)*	4	4	8
Carolina Wren *(Thryothorus ludovicianus)*	55	28	83
Cerulean Warbler *(Dendroica cerulea)*	6	6	12
Chipping Sparrow *(Spizella passerina)*	0	3	3
Common Yellowthroat *(Geothlypis trichas)*	1	4	5
Downy Woodpecker *(Picoides pubescens)*	8	8	16
Eastern Bluebird *(Sialia sialis)*	1	0	1
Eastern Phoebe *(Sayornis phoebe)*	3	30	33
Eastern Tufted Titmouse *(Baeolophus bicolor)*	14	5	19
Eastern Wood-Pewee *(Contopus virens)*	6	7	13
Field Sparrow *(Spizella pusilla)*	1	0	1
Hairy Woodpecker *(Picoides villosus)*	4	3	7
Hooded Warbler *(Wilsonia citrina)*	55	41	96
Indigo Bunting *(Passerina cyanea)*	54	72	126
Kentucky Warbler *(Oporornis formosus)*	10	7	17
Louisiana Waterthrush *(Seiurus motacilla)*	1	6	7
Magnolia Warbler *(Dendroica magnolia)*	0	1	1
Northern Cardinal *(Cardinalis cardinalis)*	10	8	18
Ovenbird *(Seiurus aurocapillus)*	51	32	83
Red-bellied Woodpecker *(Melanerpes carolinus)*	1	1	2
Red-eyed Vireo *(Vireo olivaceus)*	34	39	73
Scarlet Tanager *(Piranga olivacea)*	6	9	15
Summer Tanager *(Piranga rubra)*	2	0	2
Swainson's Thrush *(Catharus ustulatus)*	0	1	1
White-breasted Nuthatch *(Sitta carolinensis)*	3	6	9
White-eyed Vireo *(Vireo griseus)*	0	3	3
Wood Thrush *(Hylocichla mustelina)*	14	13	27
Worm-eating Warbler *(Helmitheros vermivora)*	54	57	111
Yellow Warbler *(Dendroica petechia)*	0	1	1
Yellow-bellied Flycatcher *(Empidonax flaviventris)*	2	1	3
Yellow-throated Vireo *(Vireo flavifrons)*	1	0	1
Totals	437	458	895

Table 4. Numbers of individuals banded, total abundance, species richness and Shannon's diversity index for all banding sites during the 2002 banding season.

Species	WPLF	WPCC	CCLF	CCCC	BCLF	BCCC
Acadian Flycatcher *(Empidonax virescens)*	0	0	1	1	2	0
American Goldfinch *(Carduelis tristis)*	0	0	0	0	0	0
American Redstart *(Setophaga ruticilla)*	0	1	0	0	10	6
Black-and-white Warbler *(Mniotilta varia)*	0	0	2	1	2	4
Blue-gray Gnatcatcher *(Polioptila caerulea)*	0	1	1	0	1	1
Brown-headed Cowbird *(Molothrus ater)*	0	0	0	0	1	1
Carolina Chickadee *(Poecile carolinensis)*	1	0	0	3	0	0
Carolina Wren *(Thryothorus ludovicianus)*	11	9	3	10	12	10
Cerulean Warbler *(Dendroica cerulea)*	2	0	1	0	1	2
Chipping Sparrow *(Spizella passerina)*	0	0	0	0	0	0
Common Yellowthroat *(Geothlypis trichas)*	0	0	0	0	1	0
Downy Woodpecker *(Picoides pubescens)*	2	1	1	2	2	0
Eastern Bluebird *(Sialia sialis)*	1	0	0	0	0	0
Eastern Phoebe *(Sayornis phoebe)*	3	0	0	0	0	0
Eastern Tufted Titmouse *(Baeolophus bicolor)*	2	1	1	0	8	2
Eastern Wood-Pewee *(Contopus virens)*	2	0	0	3	1	0
Field Sparrow *(Spizella pusilla)*	0	1	0	0	0	0
Hairy Woodpecker *(Picoides villosus)*	0	1	0	0	3	0
Hooded Warbler *(Wilsonia citrina)*	1	14	4	5	24	7
Indigo Bunting *(Passerina cyanea)*	10	5	7	8	15	9
Kentucky Warbler *(Oporornis formosus)*	2	3	1	2	1	1
Louisiana Waterthrush *(Seiurus motacilla)*	0	0	0	0	1	0
Magnolia Warbler *(Dendroica magnolia)*	0	0	0	0	0	0
Northern Cardinal *(Cardinalis cardinalis)*	2	0	1	0	3	4
Ovenbird *(Seiurus aurocapillus)*	5	9	15	13	8	1
Red-bellied Woodpecker *(Melanerpes carolinus)*	1	0	0	0	0	0
Red-eyed Vireo *(Vireo olivaceus)*	5	8	9	7	4	1
Scarlet Tanager *(Piranga olivacea)*	0	1	4	0	0	1
Summer Tanager *(Piranga rubra)*	2	0	0	0	0	0
Swainson's Thrush *(Catharus ustulatus)*	0	0	0	0	0	0
White-breasted Nuthatch *(Sitta carolinensis)*	1	0	0	0	0	2
White-eyed Vireo *(Vireo griseus)*	0	0	0	0	0	0
Wood Thrush *(Hylocichla mustelina)*	2	3	5	1	2	1
Worm-eating Warbler *(Helmitheros vermivora)*	0	6	9	7	12	20
Yellow Warbler *(Dendroica petechia)*	0	0	0	0	0	0
Yellow-bellied Flycatcher *(Empidonax flaviventris)*	0	0	0	1	0	1
Yellow-throated Vireo *(Vireo flavifrons)*	1	0	0	0	0	0
Total Abundance	56	64	65	64	114	74
Species Richness	19	15	16	14	21	18
Shannon's Diversity Index	1.129	1.002	1.027	1.008	1.103	1.034

Table 5. Numbers of individuals banded, total abundance, species richness and Shannon's diversity index for all banding sites during the 2003 banding season.

Species	WPLF	WPCC	CCLF	CCCC	BCLF	BCCC
Acadian Flycatcher *(Empidonax virescens)*	0	0	0	8	0	0
American Goldfinch *(Carduelis tristis)*	2	0	0	0	0	0
American Redstart *(Setophaga ruticilla)*	0	1	2	0	3	10
Black-and-white Warbler *(Mniotilta varia)*	0	9	5	1	1	3
Blue-gray Gnatcatcher *(Polioptila caerulea)*	3	0	0	0	3	1
Brown-headed Cowbird *(Molothrus ater)*	4	1	1	0	3	1
Carolina Chickadee *(Poecile carolinensis)*	2	0	0	0	1	1
Carolina Wren *(Thryothorus ludovicianus)*	4	3	2	9	3	7
Cerulean Warbler *(Dendroica cerulea)*	0	2	0	0	0	4
Chipping Sparrow *(Spizella passerina)*	0	0	3	0	0	0
Common Yellowthroat *(Geothlypis trichas)*	0	0	0	0	2	2
Downy Woodpecker *(Picoides pubescens)*	6	0	2	0	0	0
Eastern Bluebird *(Sialia sialis)*	0	0	0	0	0	0
Eastern Phoebe *(Sayornis phoebe)*	27	0	3	0	0	0
Eastern Tufted Titmouse *(Baeolophus bicolor)*	3	0	0	0	0	2
Eastern Wood-Pewee *(Contopus virens)*	4	0	0	0	2	1
Field Sparrow *(Spizella pusilla)*	0	0	0	0	0	0
Hairy Woodpecker *(Picoides villosus)*	2	0	0	0	0	1
Hooded Warbler *(Wilsonia citrina)*	0	13	4	7	6	11
Indigo Bunting *(Passerina cyanea)*	17	12	2	1	20	20
Kentucky Warbler *(Oporornis formosus)*	0	2	0	0	2	3
Louisiana Waterthrush *(Seiurus motacilla)*	1	0	3	1	0	1
Magnolia Warbler *(Dendroica magnolia)*	0	0	0	0	0	1
Northern Cardinal *(Cardinalis cardinalis)*	0	2	2	1	0	3
Ovenbird *(Seiurus aurocapillus)*	0	12	6	9	3	2
Red-bellied Woodpecker *(Melanerpes carolinus)*	1	0	0	0	0	0
Red-eyed Vireo *(Vireo olivaceus)*	7	6	2	2	6	16
Scarlet Tanager *(Piranga olivacea)*	5	0	3	0	1	0
Summer Tanager *(Piranga rubra)*	0	0	0	0	0	0
Swainson's Thrush *(Catharus ustulatus)*	0	0	0	1	0	0
White-breasted Nuthatch *(Sitta carolinensis)*	3	1	0	1	0	1
White-eyed Vireo *(Vireo griseus)*	0	2	0	0	0	1
Wood Thrush *(Hylocichla mustelina)*	1	5	3	4	0	0
Worm-eating Warbler *(Helmitheros vermivora)*	0	15	9	12	5	16
Yellow Warbler *(Dendroica petechia)*	0	0	0	0	1	0
Yellow-bellied Flycatcher *(Empidonax flaviventris)*	0	0	0	0	0	1
Yellow-throated Vireo *(Vireo flavifrons)*	0	0	0	0	0	0
Total Abundance	92	86	52	57	62	109
Species Richness	17	15	16	13	16	23
Shannon's Diversity Index	1.019	1.020	1.140	0.944	1.021	1.116

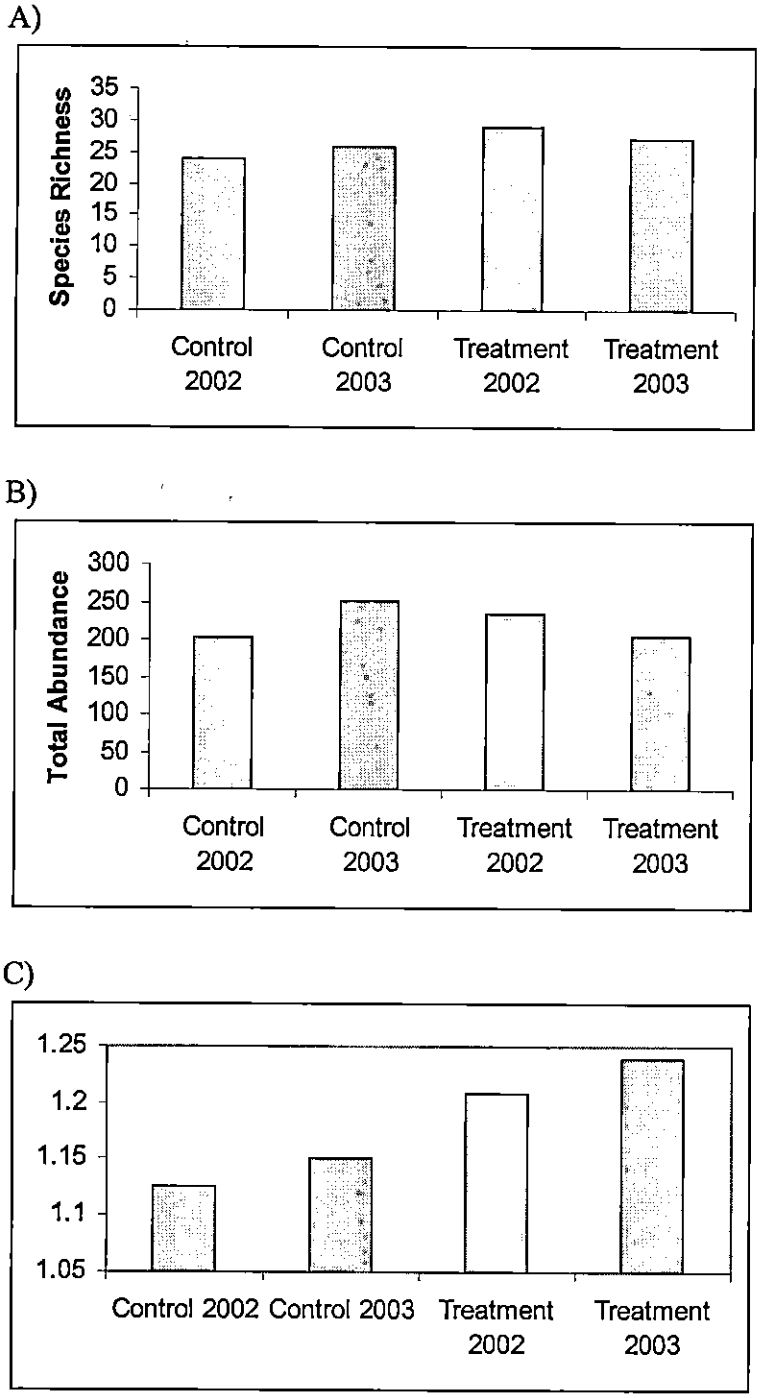

Figure 5. Species richness (A), total abundance (B), and Shannon's index (C) for control and treatment sites for 2002 and 2003.

2002 versus 2003 for both the treatment sites and control sites, though again treatment sites had lower total abundances in addition to lower species richness and diversity (Figure 5).

4.2 Habitat

For habitat analysis, percent coverage for the upperstory, midstory, understory, live ground cover, dead ground cover, and total non-vegetative cover were compared for 2003 (Table 6). There was a significant difference ($p = 0.046$) between the treatment sites and control sites for percent upperstory cover. The other variables were not significant (Table 7) though average percent cover for understory and living ground cover was greatly lower in areas that had been burned as compared to control sites.

4.3 Nesting Guilds

4.3.1 Ground Nesting

There were no significant differences between the 2002 and 2003 banding seasons for treatment sites and control sites (Table 8), though abundance of ground nesting species of treatment sites was dramatically lower in 2003 than in 2002. In comparing treatment sites to control sites there were no significant differences in 2002 (Table 10), but there was a significant differences in 2003 ($p = 0.046$) with treatment sites having fewer ground nesters than control sites (Table 9).

4.3.2 Low Shrub Nesting

There were no significant differences between the 2002 and 2003 banding seasons for treatment sites or control sites (Table 8) or for control sites versus

Table 6. Percent cover for vegetative levels for each banding site (August 2002).

Canopy Level	WPLF	WPCC	CCLF	CCCC	BCLF	BCCC
UpperStory	64	60	62	61	65	60
Midstory	34	55	50	50	48	35
Understory	10	40	32	30	29	45
Ground Cover live veg.	21	20	15	25	25	40
Ground Cover dead veg.	35	50	60	20	20	15
Total non-vegetative	15	30	25	55	60	45

Table 7. Average percent vegetative cover for treatment sites versus control sites for 2003.

Canopy Level	Treatment	Control
UpperStory	63.67	60.33
Midstory	44.00	46.67
Understory	23.67	38.33
Ground Cover live veg.	20.33	28.33
Ground Cover dead veg.	38.33	28.33
Total non-vegetative	33.33	43.33

Table 8. Comparison of total abundances for treatment sites versus control sites for six nesting guilds for 2002 and 2003.

Nesting Guilds (n = 3)	Treatment	Control	P-value
Ground Nesting			
2002	84	96	0.658
2003	44	105	0.046
Low Shrub Nesting			
2002	61	48	0.507
2003	49	82	0.275
Midstory Nesting			
2002	46	33	0.369
2003	26	58	0.046
Canopy Nesting			
2002	16	9	0.072
2003	23	10	0.184
Cavity Nesting			
2002	22	12	0.513
2003	20	7	0.513
Rock Nesting			
2002	3	0	0.317
2003	30	0	0.121

Table 9. Comparison of total abundances of individuals in each nesting guild for 2002 versus 2003 for treatment and control banding sites.

Nesting Guild	2002	2003	P-value
Treatment Sites (n = 3)			
Ground Nesting	84	44	0.127
Low Shrub Nesting	61	49	0.825
Midstory Nesting	46	26	0.105
Canopy Nesting	16	23	0.507
Cavity Nesting	22	20	0.663
Rock Nesting	3	30	0.346
Control Sites (n = 3)			
Ground Nesting	96	105	0.825
Low Shrub Nesting	48	82	0.513
Midstory Nesting	33	58	0.046
Canopy Nesting	9	10	0.658
Cavity Nesting	12	7	0.369
Rock Nesting	0	0	1.000

treatment sites for 2002 or 2003 (Table 10). For 2003, a greater number of low shrub nesting species were captured in control sites compared to treatment sites, though not significant (p = 0.275).

4.3.3 Midstory Nesting

There was a significant difference (p = 0.046) of numbers of midstory nesters between treatment and control sites for 2003, but not for 2002 (Table 8). Control sites had more captured midstory nesting species throughout the 2003 season. Control sites saw a significant increase (0.046) in the number of captured midstory nesters, while no significant differences were seen in the treatment sites (Table 9).

4.3.4 Canopy Nesting

No significant differences were found in comparing treatment sites to control sites for 2002 or 2003 (Table 8). However, there were fewer canopy nesting species captured in the control sites compared to the treatment sites (mean abundance 3.3 vs. 7.7) following the burn in 2003. There were no significant differences in abundance of canopy nesters for the 2002 versus 2003 nesting seasons for treatment or control sites (Table 9).

4.3.5 Cavity Nesting

For cavity nesting species there were no significant differences found in treatment sites versus control sites (Table 8). Treatment sites in both banding seasons on average had more captured cavity nesting species than that of control sites. No differences were found in comparing the 2002 versus the 2003 seasons for control or treatment sites (Table 9).

4.3.6 Rock Nesting

There was only one rock nesting species captured in both banding seasons (Table 8). Following the spring burn in 2003, treatment sites had more captured rock nesting individuals (n = 10) than treatment sites (n = 1); however this difference was not significant (Table 9). No differences were found in comparing the 2002 versus the 2003 seasons for control or treatment sites (Tables 10).

4.4 Foraging Guilds

4.4.1 Ground Foraging

In comparing the treatment sites to the control sites for 2002 and 2003, there were no significant differences in the abundance of ground foraging species (Table 10). Mean abundances were substantially lower in treatments sites for the 2003 season. In comparing the 2002 versus the 2003 seasons, there was a significant difference in the treatment sites, with a significantly lower number ($p = 0.043$) of ground foraging species following the burn (Table 11). No significant differences were found in the 2002 versus the 2003 seasons for control sites (Table 11).

4.4.2 Low Shrub Foraging

There were no significant differences in the abundance of low shrub foraging species before and after the burn in 2003 (Table 10) in comparing treatment sites and control sites. Abundances were substantially lower in treatments sites in the 2003 season. In comparing the 2002 versus the 2003 seasons, there were no significant differences between the treatment sites and the control sites (Table 11).

Table 10. Comparison of avian total abundances for treatment sites versus control sites for six foraging guilds for 2002 and 2003.

Foraging Guilds (n = 3)	Treatment	Control	P-value
Ground Foraging			
2002	64	57	0.827
2003	26	53	0.121
Low Shrub Foraging			
2002	86	87	0.513
2003	65	42	0.127
Midstory Foraging			
2002	34	27	0.658
2003	22	44	0.513
Canopy Foraging			
2002	13	9	0.261
2003	17	10	0.376
Bole Foraging			
2002	14	11	0.376
2003	20	17	0.827
Aerial Foraging			
2002	9	4	0.268
2003	36	9	0.275

Table 11. Comparisons of total avian abundances of six foraging type guilds between 2002 and 2003 for treatment and control sites.

Foraging Guild	2002	2003	P-value
Treatment Sites (n = 3)			
Ground Foraging	64	26	0.043
Low Shrub Foraging	86	65	0.827
Midstory Foraging	34	22	0.184
Canopy Foraging	13	17	0.500
Cavity Foraging	14	20	0.658
Rock Foraging	9	36	0.658
Control Sites (n = 3)			
Ground Foraging	57	53	0.513
Low Shrub Foraging	87	42	0.513
Midstory Foraging	27	44	0.658
Canopy Foraging	9	10	0.658
Cavity Foraging	11	17	0.658
Rock Foraging	4	9	0.487

4.4.3 Midstory Foraging

For midstory foraging species there were no significant differences pre- and post-burn for control sites versus treatment sites and for the 2002 season versus the 2003 banding season for control and treatment sites (Tables 10 and 11). Overall abundances for midstory foraging species did decrease in 2003 following the burn for treatment sites.

4.4.4 Canopy Foraging

No significant differences or trends were found in control sites versus treatment sites for 2002 or 2003 nor were there differences found in 2002 versus 2003 for control and treatment banding sites (Tables 10 and 11). Low numbers of canopy species were captured throughout the study due to the position of the mist nets.

4.4.5 Bole Foraging

Bole foraging species abundances were higher (but not significant) in treatment sites than control sites before and after the burn (Tables 10 and 11). There were no significant differences in bole foraging species abundance between treatment sites and control sites.

4.4.6 Aerial Foraging

Aerial foraging species increased in 2003 following the burn in treatment sites, but no significant differences were found in control versus treatment sites for 2002 and 2003 nor were there differences found in 2002 versus 2003 for control and treatment banding sites (Tables 10 and 11).

4.5 Brown-headed Cowbird Populations

In treatment sites the average number of Brown-Headed Cowbirds captured increased from 0.333 birds per site in 2002 to 2.667 birds in 2003. This increase was not significant, but was unlike cowbirds numbers in control sites which did not change from 2002 to 2003 (Table 12). There were no significant differences between treatment and control sites for either 2002 or 2003 but cowbird numbers were greater in treatment sites in 2003 compared to control sites (Table 13).

Table 12. Average mean abundance of Brown-Headed Cowbirds captured in treatment sites versus control sites for 2002 and 2003.

Year	Treatment	Control	P-value
2002	0.333	0.333	1.000
2003	2.667	0.667	0.105

Table 13. Average mean abundances of Brown-Headed Cowbirds captured in 2002 versus 2003 for treatment and control sites.

Banding Site	2002	2003	P-value
Treatment	0.333	2.667	0.072
Control	0.333	0.667	0.456

CHAPTER V

Discussion

The breeding bird community of the Pioneer Weapons Area did not significantly change following the single prescribed burn in the spring of 2003. Even though the prescribed burn appeared to have no statistically significant effect on the total number of breeding birds, the total number of bird species present decreased in treatment sites following the burn. While the diversity of birds increased following the burn. This trend in treatment sites is not necessarily represented in the statistical analyses used due to the small sample size (n = 3). Total abundances for both banding seasons remained consistent, with at least 200 individuals captured each year for both control sites and treatment sites (Figure 5). In addition no single guild of bird species was eliminated from any of the treatment sites during the course of the study.

The burn appeared to have some negative and positive affects on individual guilds of birds present in the study area. Because the fire was concentrated in the lower canopy levels of the forest, birds that utilize the forest floor and shrub layers were the most affected. Ground nesting and foraging species (Tables 1 and 2) such as the Wood Thrush, Ovenbird, Worm-eating Warbler, and Carolina Wren, which were among some of the most numerous species captured (Tables 3 - 5), all decreased in abundance following the 2003 burn.

Ovenbirds and Carolina Wrens utilize leaf litter for building and concealment of their nest in the spring of the year (Horn and Donovan, 1994; Haggerty and

Morton, 1995). Surface fires from the prescribed burns decreased or the leaf litter from the treatment sites following the burn. Worm-eating Warblers tend to utilize small, open depressions in forested stream bottoms with high moisture content (Hanners and Patton, 1998). The burn plot set by the U.S. Forestry Service utilized streams as fire breaks; so much of the suitable habitat utilized by Worm-eating Warblers was subject to being burned by the fire. Fire potentially could have altered the moisture content of the soil by the overall reduction of leaf litter and increasing the amount of sunlight cast upon the exposed soil (Artman et al., 2001).

All of these species (Tables 1 and 2) are insectivores and hunt insects by foraging in leaf litter or in thick vegetation close to the ground (Sibley, 2000). The introduction of fire greatly reduced the quantity of living vegetation and the amount of leaf litter, potentially destroying the prey along with the habitat. With no suitable habitat and food available birds that utilize the ground for nesting, foraging or both would have to move outside the areas burned by the fire in order to find suitable habitat.

Low shrub nesting species, such as the Hooded Warbler and the Indigo Bunting, typically nest less than one meter from the ground in thick brush (Ogden and Stutchbury, 1994; Payne, 1992). Following the burn the majority of shrubs (spicebush, sassafras, and red maple) were eliminated and regeneration did not occur until after fledging. Hooded Warblers decreased the most, but both Hooded Warblers and Indigo Buntings were not captured in great numbers, if at all, until the sassafras shrub layer and herbaceous vegetation regenerated later in the summer (Figure 6). Indigo Buntings ultimately increased by seven in 2003 compared to 2002 (Tables 4 and 5). Indigo Buntings feed on a variety of foods including berries and seeds (Payne, 1992).

Fruits growing on vegetation that regenerated later in the season as a result of the fire probably played a key role in attracting Indigo Buntings to areas burned in search of food. Insectivores, such as Hooded Warblers, appeared not to benefit as much from the vegetative regeneration.

Midstory nesting species did decline following the burn in treatment sites with only one species, Red-eyed Vireo, being caught in large numbers. Many midstory nesting species captured were caught in low numbers, making it difficult to assess with confidence the overall affects from the burn. In some treatments sites, the fire burned at a greater temperature, causing vegetation several meters off the ground to be affected. This affect combined with the damage to the ground surface possibly played a role in reducing the number of prey species available to these members of this bird guild, which primarily forage from the midstory down to the forest floor in search of insects and other invertebrates.

The design of the mist nets severely impairs the capture rate of birds that do not fly or forage in the lower canopy levels. During the course of the study very few canopy nesting and foraging individuals (Tables 1 and 2) were captured. Nets that did capture canopy species were typically located in higher elevations. The limited data did not demonstrate that fire had an effect on the species. The short duration of

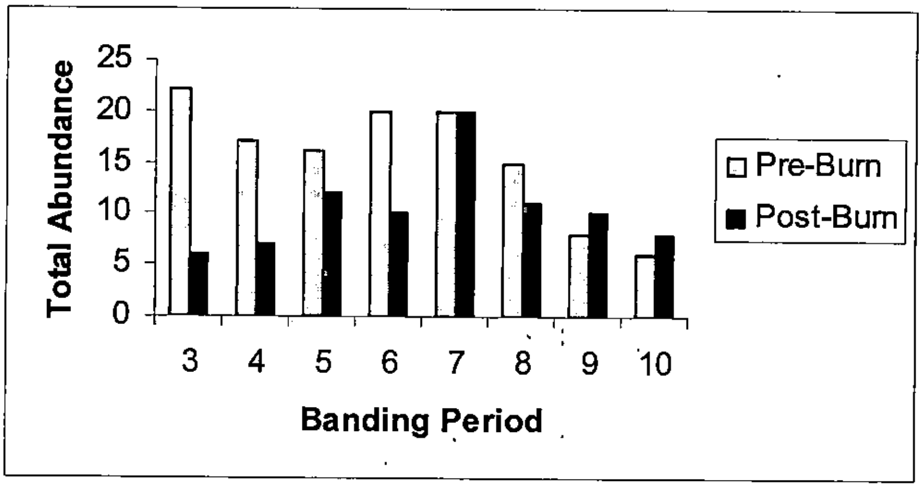

Figure 6. Bar graph of total abundance of Hooded Warblers and Indigo Buntings for the 2002 and 2003 banding seasons for burned sites.

the burn mostly affected vegetation less than one meter off the ground, leaving larger trees inhabited by canopy bird species unaffected. Long term repeated burning could potentially lead to shifts in tree species composition, which could potentially effect upper canopy species.

Cavity nesting species, which were caught in low numbers due to the low position in the forest canopy of the mist nets, increased in both treatment sites and control sites. Since the prescribed burn mostly affected small trees, shrubs, and herbaceous vegetation, larger trees in which these species nest would not have been affected (Lanham et al., 2002). In February of 2003 a severe ice storm hit the eastern half of Kentucky, including the study site, resulting in a large number of trees that fell in response to the weight of ice. The ice storm hit before the scheduled prescribed burn making it hard to distinguish whether the small increase in cavity nesting and bark foraging species is a response to the prescribed burning or the ice storm. Two separate studies (Faccio, 2003; Blais et al., 2000) looked at the affects of ice storms on forest bird communities. Both found that most bark foraging species were unaffected from an initial ice storm. In fact one species, the Downy Woodpecker, even decreased significantly following the storm in one of the studies (Blais et al. 2000). Both studies utilized point count methods and were not targeting bark foraging species, making unclear whether increased amount of decaying wood benefited these species as expected. Due to the low numbers of individuals of these guilds captured during the study it cannot be determined whether either disturbance ultimately had any affects on this specific guild of bird species.

Only one guild of species appeared to be positively affected by the burn. Aerial foraging species, though caught in low numbers, tended to substantially increase in areas that had been burned. As a result of the fire, vegetation up to three meters was void of leaves for the majority of the banding season (personal observation, 2003). The lack of vegetation tended to favor aerial foraging species, which capture insects on the wing from open perches in mid-canopy (Weeks, 1994). Eastern Phoebes, which represented the only rock nesting species captured, increased following the burn. Eastern Phoebes benefited from the abundance of rock, previously covered by vegetation, exposed following the burn. WPLF, which contained more rock than the other treatment sites, had the greatest increase (24 individuals) Eastern Phoebes following the burn.

Brown-headed Cowbirds appeared to be positively affected from the opening of the forest from the burns. The numbers of cowbirds captured increased in burned areas in 2003, in addition to individuals visually observed at sites that had burned (unpublished data). The openness of the forest following the burn could have increased the vulnerability of many bird species to the brood parasitism from Brown-headed Cowbirds.

Even with the burns all taking place in the spring of 2003, weather delays between the burning of individual burn plots created some difference in the condition of the forest when the burn was initiated. BCLF and WPLF were burned after leaf development in the lower and upper canopies but CCLF was burned before leaf development. In areas that had green leaves, the fire to burned at a higher temperature resulting in a higher degree of destruction within the leaf litter, shrubs, forbs, and lower canopy trees. The habitat with out leaves did not burn as hot and resulted in a heterogeneous environment with burned and non-burned patches of habitat. The combination of burned and non-burned patches of habitat played a key role in the stability of ground nesting and shrub nesting birds throughout the banding season. The difference can be seen in the stronger effects on overall abundances of ground and low shrub nesting captured species in CCLF compared to WPLF (Figure 7).

Results of this study are consistent with the findings of other studies in mixed-oak forests in the eastern U.S. in that no overall significant changes in the breeding bird community were found following a single prescribed burn. In a study by Artman et al. (2001), out of the 30 bird species captured in their study, ground and low shrub nesting species were the most negatively affected following a burn, while aerial foraging species increased in abundance. A similar study by Aquilani et al. (2000) found abundances of ground and shrub nesting species to be greater in non-burned areas as compared to burned. The greatest difference in abundance they found was for the Ovenbird, a ground nesting species. They also found that species nesting in recently burned areas were of greater risk from brood parasitism by the Brown-headed Cowbird.

Data collected suggest that even though prescribed burns do not initially eliminate specific guilds of birds species, ground and low shrub nesting species may be

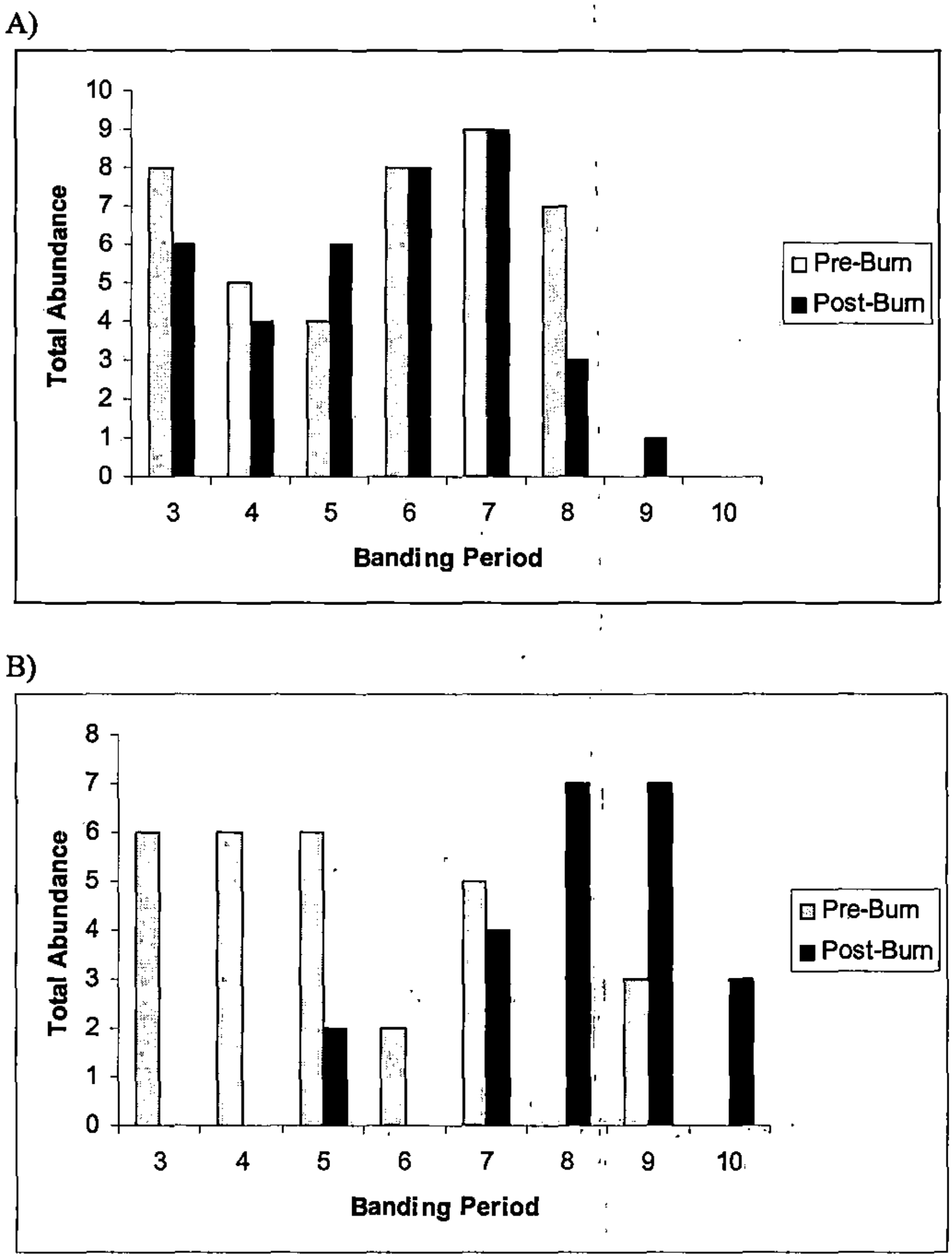

Figure 7. Bar graph of total abundance of ground and low shrub nesting species for the 2002 and 2003 banding seasons for CCLF (A) and WPLF (B).

adversely impacted as a result of the lost vegetation cover on the ground and in the shrub layers. Since the study only looks at the effects on birds immediately after the burn, it is hard to consider at the ultimate impact that prescribed burns would have on forest dwelling birds.

Future fires could potentially lead to the intended purpose of the restoration of oak seedlings, possibly filling the void left by the destruction of fire-intolerant species. The frequency of the burns would have the most impact on the forest community. Frequent fire intervals could kill larger trees potentially leading to devastating vegetation shifts in the forest structure itself. The shift could result in changes to the bird community at upper canopy levels in addition to ground and shrub levels. Future studies with prescribed burns that mimic natural fire cycles of 10 to 20 years in the eastern U.S. (Blankenship and Arthur, 1999) will be needed in order to determine the time intervals between burns for forest recovery as well as the recovery and future conservation of bird species inhabit the forests.

CHAPTER VI

Conclusions

It is clearly established that forest bird communities throughout the eastern U.S. have declined over the past 100 years (Hagan and Johnston, 1989; Martin and Finch, 1995). Prescribed burning is being tested by forestry managers as a restoration tool for mixed hardwood forests. Data from this study suggest prescribed burning will not eliminate or alter populations of all species of a forest bird community following a single burn but instead will affect bird species of a specific guild level. Because most prescribed burns, including this one, affect vegetation levels mostly below one meter, bird species that utilize leaf litter and dense shrubs will be most affected. This effect is likely to be of short duration, unless burns are conducted at high frequencies. High frequency burns have the potential to change the landscape from a forested habitat to a more open savanna-like habitat destroying fire-intolerant species, such as oaks, that have adapted to less frequent fires.

Upper canopy foraging and nesting species, though captured in low numbers compared to lower canopy species, were relatively unaffected from the burn due to the larger trees they inhabit being relatively unaffected from the surface fires. Aerial foraging species increased in treatment sites following the 2003 burn. The lack of vegetation created by the fire provided ideal open foraging habitat for these species, especially the Eastern Phoebe, which increased more than any other species in the study following the burn. Long-term effects on these species may be different, because continued burning is likely to alter upper canopy species composition and densities. Changes in the percent coverage of canopy layers from continued burning could possibly play a vital role in upper canopy species survival and composition (Rodewald and Smith, 1998; Robdwald and Abrams, 2002).

Prescribed burning is a practical method of reversing the affects of decades of fire suppression in forests. Though the use of controlled burns has been shown capable to destroy fire unwanted fire-intolerant species, the affects of the fire on the animal species that inhabit the forest has been relatively unstudied. Burning on a frequent basis could lead to vegetation shifts in birds species that inhabit the leaf litter to upper canopy bird species. Sufficient recovery time for bird and other animal species must be studied in order to determine adequate burn frequencies, as well as burn intensities, that pose minimal harm to the animal fauna but yet can control unwanted tree and shrub species expansion.

CHAPTER VII

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Figure A1. Aerial photo of mist net locations for the Buck Creek Treatment (BCLF) Site.

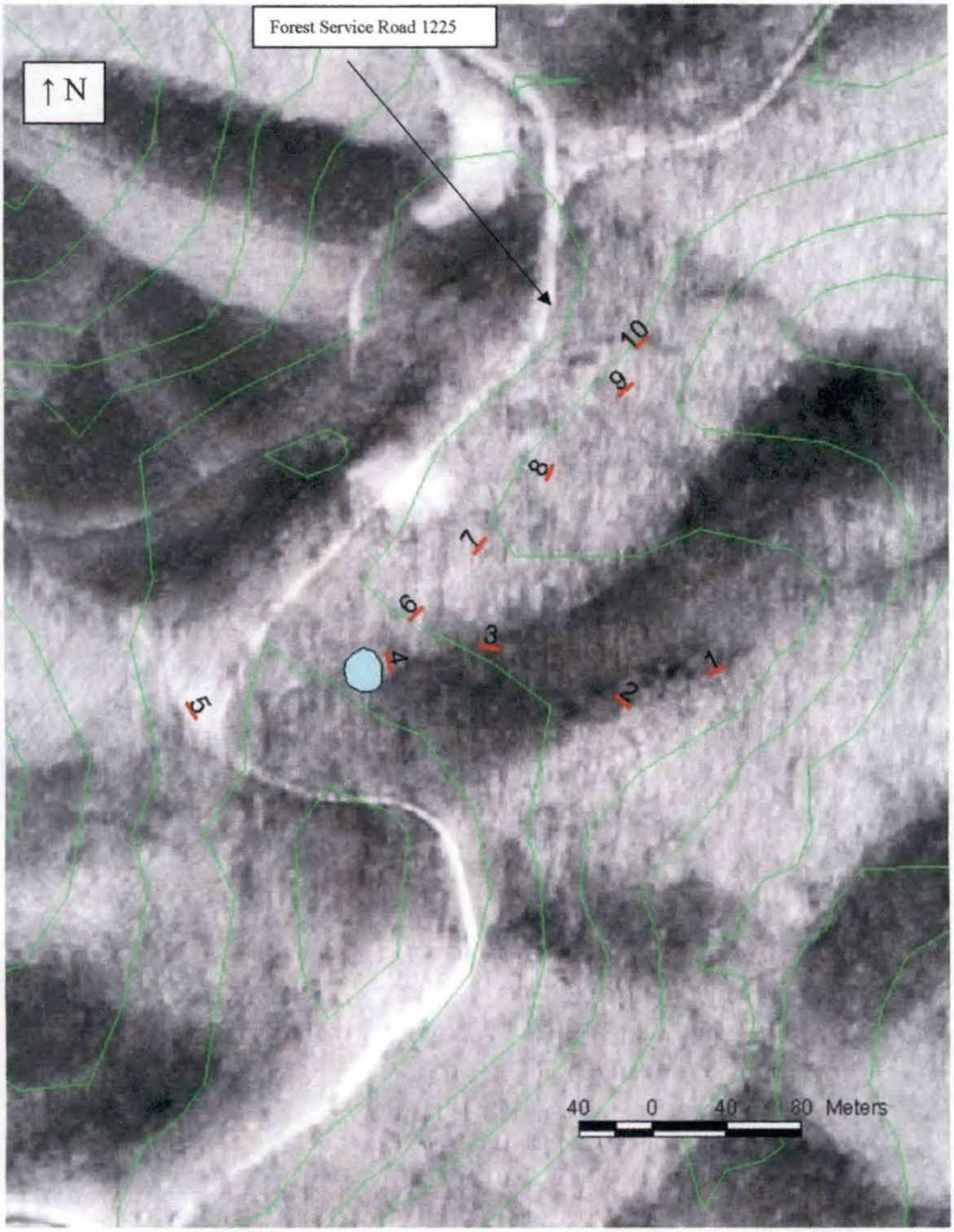

Figure A2. Aerial photo of mist net locations for the Buck Creek Control (BCCC) Site.

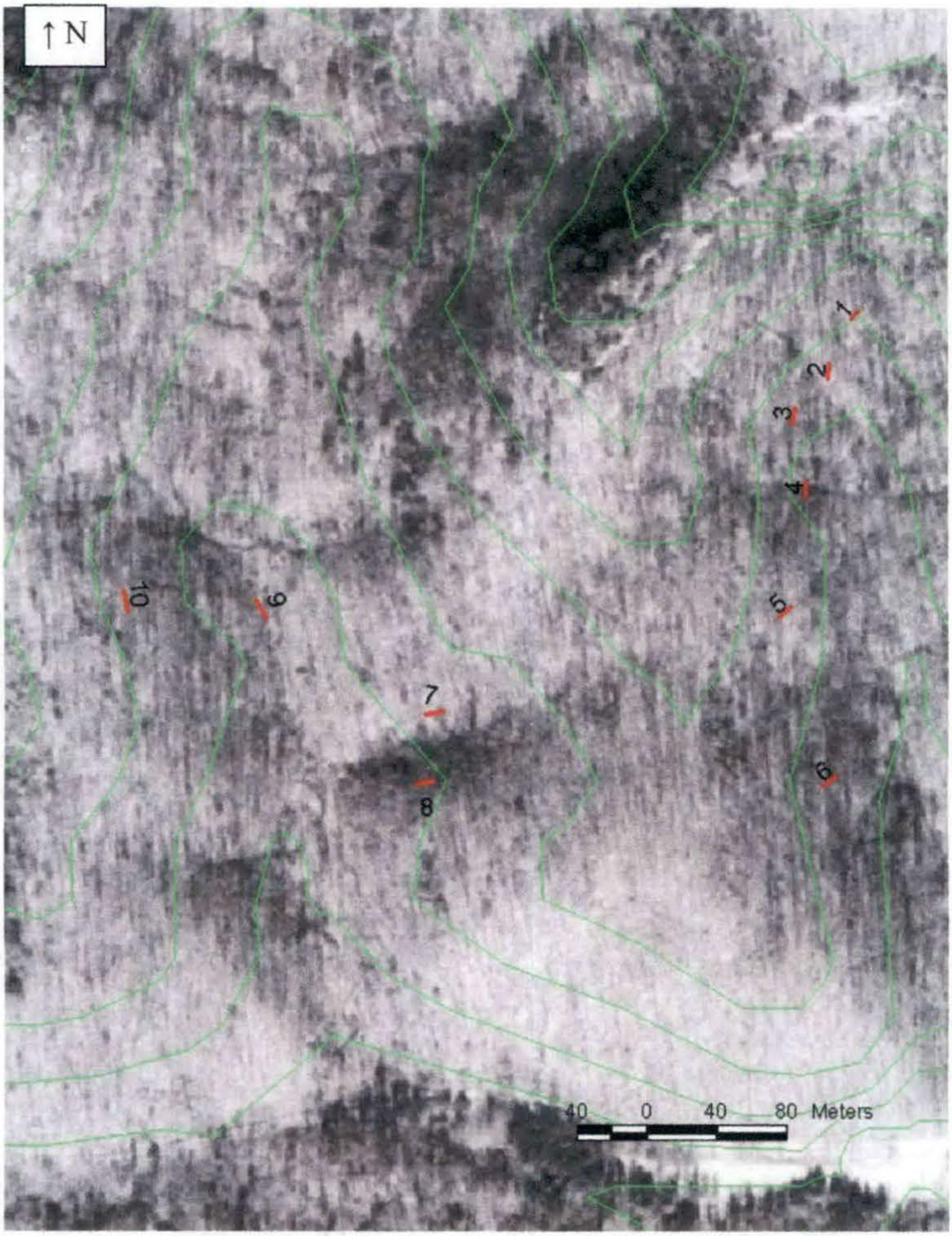

Figure A3. Aerial photo of mist net locations for the Chestnut Cliff Treatment (CCLF) Site.

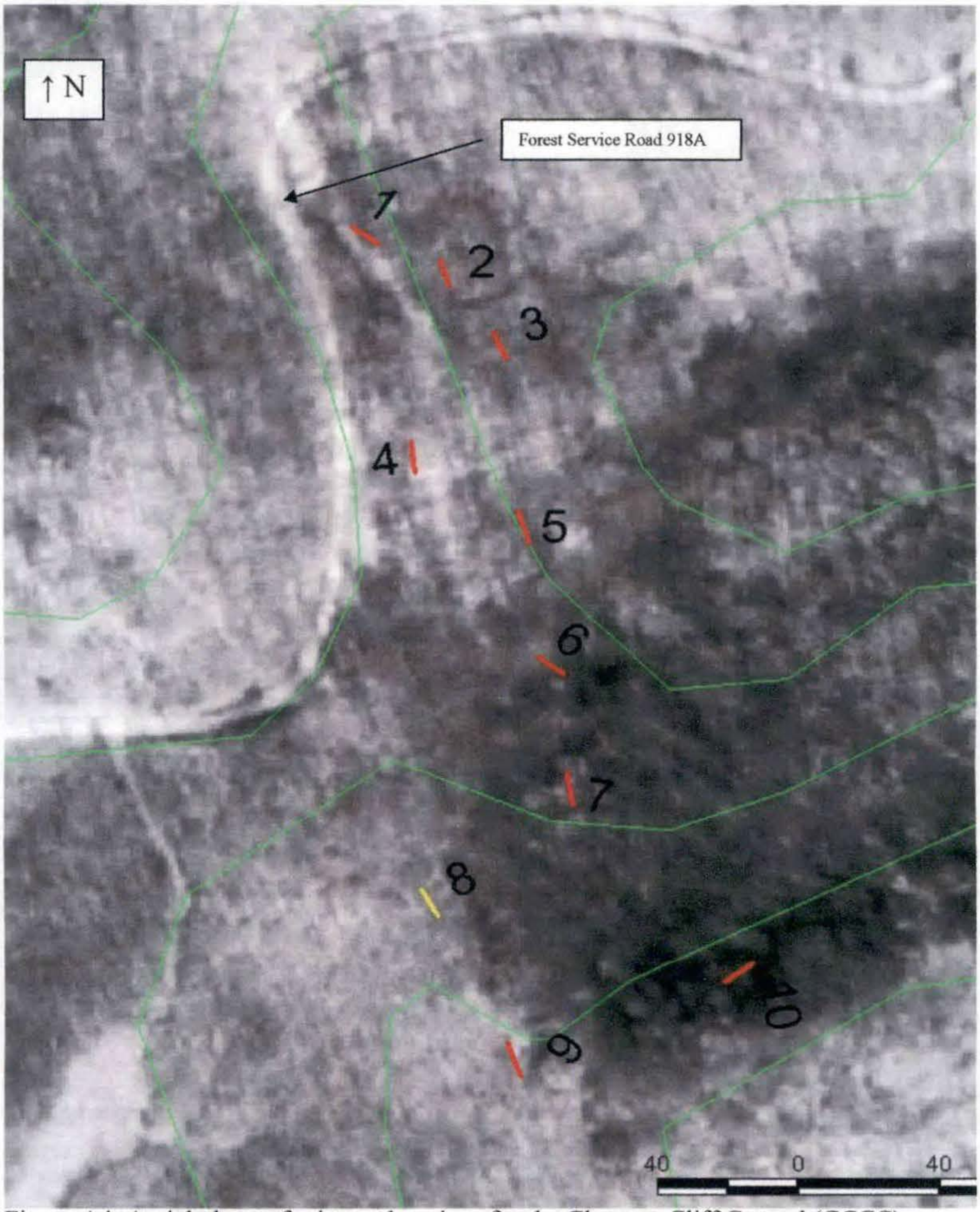

Figure A4. Aerial photo of mist net locations for the Chestnut Cliff Control (CCCC) Site.

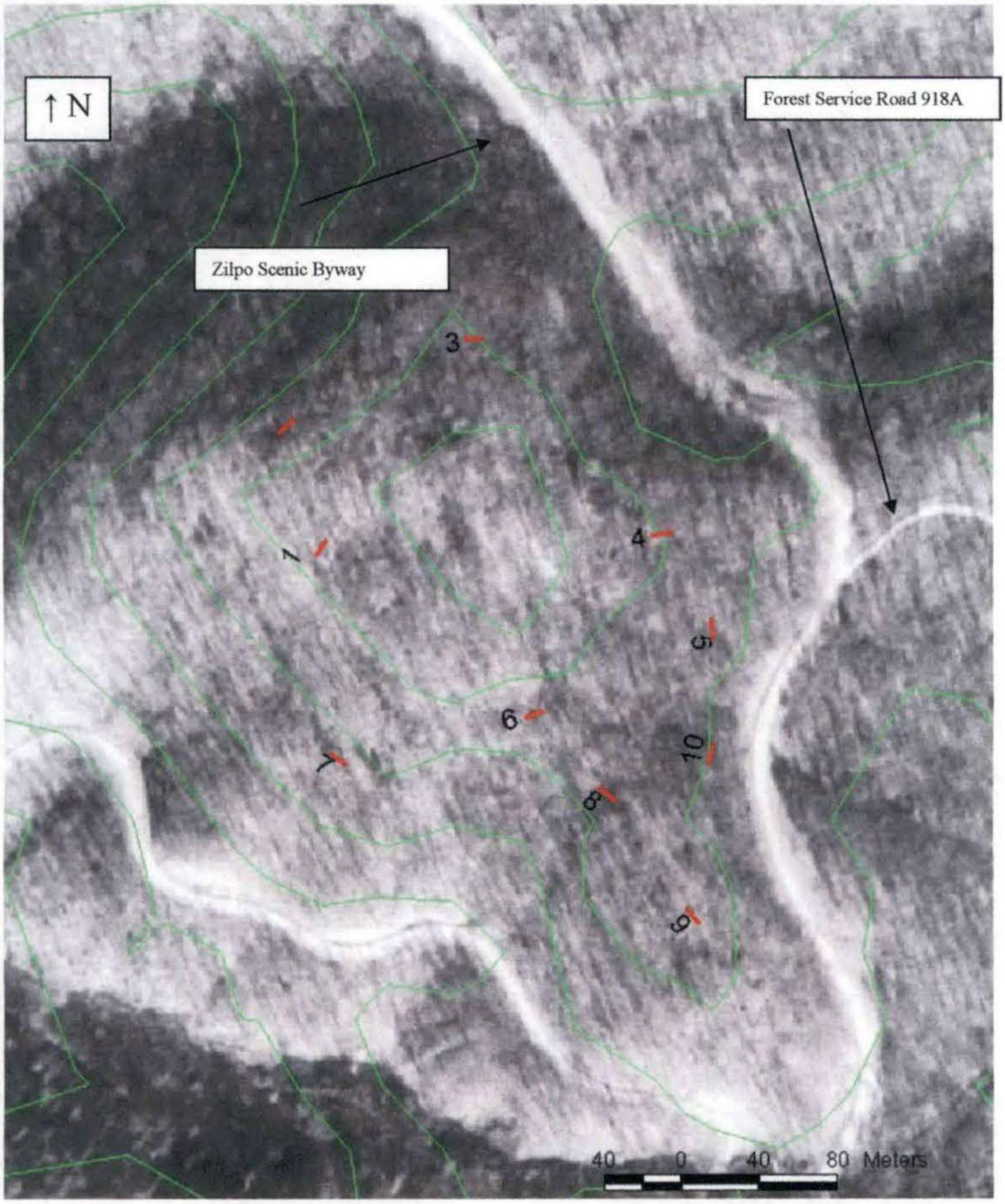

Figure A5. Aerial photo of mist net locations for the Wolfe Pen Treatment (WPLF) Site.

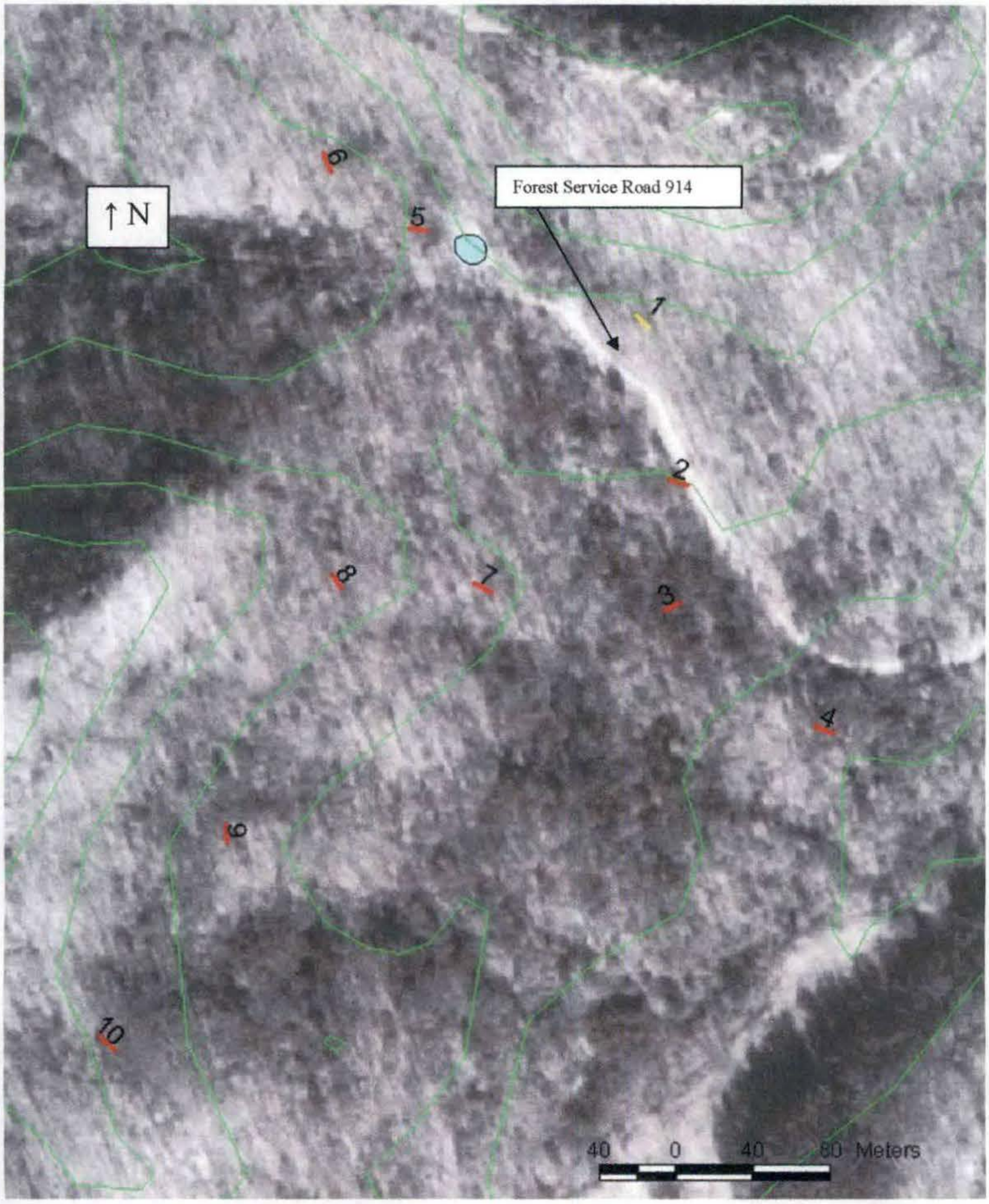

Figure A6. Aerial photo of mist net locations for the Wolfe Pen Control (WPCC) Site

APPENDIX II

Table A1. Numbers of unbanded individuals for the 2002 and 2003 banding seasons.

Species	2002	2003	Total
Acadian Flycatcher *(Empidonax virescens)*	0	1	1
American Crow *(Corvus brachyrhynchos)*	2	0	2
American Redstart *(Setophaga ruticilla)*	1	0	1
Blue Jay *(Cyanocitta cristata)*	0	1	1
Carolina Wren *(Thryothorus ludovicianus)*	1	1	2
Downy Woodpecker *(Picoides pubescens)*	0	1	1
Hooded Warbler *(Wilsonia citrina)*	1	1	2
Indigo Bunting *(Passerina cyanea)*	0	3	3
Ovenbird *(Seiurus aurocapillus)*	1	1	2
Pileated Woodpecker *Dryocopus pileatus*	1	2	3
Ruby-throated Hummingbird (*Archilochus colubris)*	15	22	37
White-breasted Nuthatch *(Sitta carolinensis)*	0	1	1
Wood Thrush *(Hylocichla mustelina)*	0	1	1
Worm-eating Warbler *(Helmitheros vermivora)*	0	1	1
Totals	22	36	58