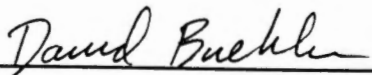
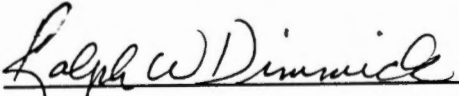
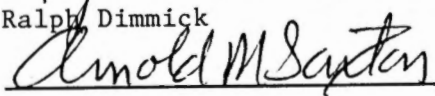


To the Graduate Council:

I am submitting herewith a thesis written by Nathan Adam Klaus entitled "Effects of forest management on songbird habitat on the Cherokee National Forest, with a special emphasis on golden-winged warblers." I have examined the final copy of this thesis for form and content and recommend that it be accepted in partial fulfillment of the requirements for the degree of Master of Science, with a major in Wildlife and Fisheries Science.


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**EFFECTS OF FOREST MANAGEMENT ON SONGBIRD HABITAT ON THE
CHEROKEE NATIONAL FOREST, WITH A SPECIAL EMPHASIS ON
GOLDEN-WINGED WARBLERS**

A Thesis
Presented for the
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Nathan A. Klaus
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ABSTRACT

The U. S. Forest Service is required under the National Forest Management Act of 1976 to consider how their management activities will sustain wildlife populations. Decreases in timber harvest and changes in the regeneration systems of Cherokee National Forest will change the availability of habitat of songbirds. I used a computer simulation of the forest to model the effects of these changes upon habitat availability for six songbird species: Acadian flycatcher (*Epidonax virescens*), blue-headed vireo (*Vireo solitarius*), chestnut-sided warbler (*Dendroica pensylvanica*), tufted titmouse (*Parus bicolor*), yellow-billed cuckoo (*Coccyzus americanus*), and yellow-throated warbler (*Dendroica dominica*). The SAS-based forest simulation model included elements of natural disturbance and variable harvest intensities. Disturbance regimes were based upon a combination of published studies, historical records in our study area, and expert opinion. The forest simulation model created a stand inventory for the Tellico, Hiwassee, and Ocoee districts of Cherokee National Forest at ten year intervals. Bird habitat models were applied to these stand inventories to quantify changes in habitat across four forest simulations of varying harvest intensity between 1993 and 2053: no-cut, expected harvests (based upon expert opinion), twice the expected harvest intensity, and three times the expected harvest intensity.

Late succession species showed substantial increases in habitat availability from 1993 to 2053. Acadian flycatcher and yellow-throated warbler habitat increased by 164

and 119%, respectively, at expected harvest levels. Blue-headed vireo and tufted titmouse habitat increased by 31 and 1.3 %, respectively, at expected harvest levels. Changes in habitat availability for these four species were inversely related to harvest intensity, however only tufted titmouse showed a decline in habitat availability at the highest harvest intensity. Based upon known breeding densities, none of these species' populations dipped below a minimum population of 250 breeding pairs. Chestnut-sided warbler remained relatively constant at expected harvest levels, but declined at lower harvest intensities. Yellow-billed cuckoo habitat declined by 21 % at expected harvest levels. Habitat for these species was directly related to the harvest intensity. Neither of these species' populations are expected to decrease below estimated minimum viable population levels (250 pairs).

Changes in timber harvest regimes may affect more disturbance-dependent species, like the golden-winged warbler. Golden-winged warbler populations have shown significant declines in many regions eliciting concern among conservationists. We measured productivity and characterized the territory and nesting habitat characteristics of golden-winged warblers in regeneration areas in the Nantahala National Forest, North Carolina, and Cherokee National Forest, Tennessee, in 1997 and 1998. Golden-winged warblers occupied young stands with low basal area ($P = 0.0310$), and better roads ($P = 0.0332$), and nested in sites with high herbaceous density ($P = 0.0114$). Nest sites had fewer saplings ($P = 0.0228$) and lower canopy cover ($P = 0.0327$) than the surrounding territory. Overall nest success was 72.5 %. Nests fledged an average of 3.65

young. Recently harvested stands (age ≤ 13 years) provided habitat for golden-winged warblers capable of supporting reproduction at levels equal to or greater than other study sites across the range of this species.

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PART ONE

**A COMPARISON OF MANAGEMENT ALTERNATIVES ON
SONGBIRD BREEDING HABITAT IN THE CHEROKEE
NATIONAL FOREST**

INTRODUCTION

Controversy surrounds the management of national forests. The United States Forest Service (USFS) spends approximately \$5,018,000 annually on lawsuits brought against their timber proposals (USFS 1997). At issue is the fundamental question of what role timber management should play in the management of national forests. Frequently negative impacts on certain wildlife species are cited as justification for decreasing or prohibiting timber harvests on public land (Harwood 1997, Mezza 1997). Non-game birds are often cited as a group negatively impacted by timber harvests (Conner et al. 1979).

Many studies have investigated the effects of timber management on bird communities. Various conclusions have been reached, many contradictory, depending on which species or group of species was being studied (Conner and Adkisson 1975, Conner et al. 1979, Dickson and Segelquist 1979). Most studies have found differences in bird communities before and after harvests. However, these differences are not always negative (Crawford et al. 1981). Along with decreases in some species of birds, increases in other species of birds typically follow timber harvests (Webb et al. 1977, Crawford et al. 1981, Thompson et al. 1992). This suggests that timber harvests may be a viable form of management for some songbirds.

More hectares of national forest are altered by timber harvest activities than by wildlife management by an order of magnitude. In 1984, a fairly typical year in the last two decades, 2,470 ha were harvested, thinned, or otherwise managed in Cherokee National Forest (CNF), while 315 ha were managed specifically for wildlife (USFS

1986). Additionally, timber harvests often yield a profit whereas the expense of wildlife management activities may not be directly recovered. Because of tightening budgets, timber harvests may continue to exceed wildlife management activities in total dollars spent, available manpower, and acreage. Therefore, management of bird communities through timber management is an attractive alternative to other management activities directed solely at wildlife.

Disturbances, including silvicultural activities, have long played an important role in shaping forest communities (Smith et al. 1996). Forest diversity is often augmented by disturbance (Whitmore 1982), and many species of birds utilize recently disturbed areas. Historic disturbances are a significant factor in determining the plant and animal communities present on a site. Tree mortality diversifies a forest by creating stands of a variety of ages and origins. This, in turn, affects wildlife use, which responds to the presence or absence of disturbance-oriented plant communities, snags, downed timber, tree species composition, and a myriad of other factors. With different disturbance histories, two stands of similar site quality may look entirely different for centuries, both in their plant and animal composition. Barring further disturbance, these stands may eventually converge on similar species composition; however, during the decades that follow such disturbances, these stands will be quite distinct, both from a timber and wildlife management view (Conner et al. 1979, Crawford et al. 1981).

Bird communities are strongly correlated with available habitat (MacArthur and MacArthur 1961, MacArthur et al. 1962, James 1971, Morrison 1981, Rice et al. 1984, Weins 1992). Vegetation structure, which is largely determined by site quality and the historic disturbance of the site, is ecologically important to birds in many ways, such as

providing display perches, shelter and nest sites, and suitable foraging habitat (Morrison 1982, Weins 1992). Using an ecosystem approach to forest management, bird populations may be effectively managed through silvicultural prescriptions by providing life requisites and augmenting limiting factors (Crawford et al. 1981, Wear et al. 1996).

Bird species turnover tracks plant species turnover as a stand moves through succession (Odum 1950, Johnston and Odum 1956, Crawford et al. 1981). Harvests set back vegetation succession, resulting in an increase in early succession bird species richness and abundance. Forest-interior species disappear from a stand after harvest and do not appreciably use the stand until it has matured (Webb et al. 1977, Conner et al. 1979, Thompson et al. 1992). While clearcutting sets back a stand's natural succession, uneven-aged silvicultural methods may accelerate natural succession, providing late-seral stage habitat for birds which might be adversely affected by clearcutting. The removal of individual trees or groups of trees favors the regeneration of shade-tolerant tree species typically found in older stands. The removal of single trees or small groups may diversify the midstory and understory, as well as the canopy by simulating natural tree mortality typical of old growth stands (Annand and Thompson 1993, Thompson 1993, Baker et al. 1996). Regeneration of shade-tolerant tree species and a diverse mid and understory are components of uneven-aged management and late succession bird habitat; however, little is known at this time about the potential of uneven-aged silviculture to provide late succession bird habitat. Uneven-aged silviculture has been little practiced until recently, leaving few opportunities to gauge its potential as a management tool (Annand and Thompson 1993).

CNF even-aged management (averaged over the last 10 years) consists of relatively small clearcuts averaging 10 ha (L. Mitchell, pers. comm). The USFS expects to reduce its use of clearcutting and begin using uneven-aged management in many harvests (R. Fryar, pers. comm). As a result, the future vegetative structure of the forest as well as the future distribution and abundance of birds is unknown. Many species of wildlife are highly dependent upon periodic disturbances such as clearcutting to provide habitat (Litvaitis 1993). The evaluation of these changes in harvest regimes and their impacts on avian richness and abundance is critical. Such a study may ensure viability of avian populations for the future as required under the National Forest Management Act. In addition, such a study may alert managers of possible shortcomings in their management scheme and will promote the conservation of biological diversity.

Underlying such a study are complex issues of minimum viable population levels (MVP). An MVP is the size of a population which will ensure a specified probability of existence over a specified period of time (demographic MVP), or a maximum rate of loss of genetic variation (genetic MVP) over a given period of time (Ewens et al. 1987:59-68). Inherent in the conservation of any species is the risk of extinction. The probability of a species' extinction is inversely related to population levels, making the maintenance of viable population levels, and thus adequate habitat, critical to the long-term survival of any species. The combination of computer simulations and real world verification has provided conservationists with speculative guidelines for MVP (Jones and Diamond 1976, Ewans et al. 1987:59-68, Pimm and Redfearn 1988). Although there is no single magic population size that guarantees the persistence of a species, estimates of 500 randomly breeding individuals provide a reasonable goal for a "typical" species (Soulé

1980:151-170). Researchers have been quick to warn that no number will guarantee a species survival since extinction is probabilistic. Such a MVP was proposed to provide a reasonable chance of survival (i.e. 95%) over the medium future (100 years or more). Recent findings suggest that MVP levels of 10,000 or more individuals may be necessary to maintain the genetic diversity to allow evolutionary viability (Lande 1995). In spite of the uncertainty, Partners In Flight (PIF) has set MVP guidelines of 250 pairs for most bird species. A minimum viable population of 500 pairs is recommended in heavily fragmented settings and 5,000 pairs in very isolated settings (C. Hunter, pers. comm).

The goal of our study was to evaluate the effects of different intensities of timber harvest upon the quantity and quality of habitat available for six bird species. Working under the assumption that these species represented the general health of most bird species in CNF, we hoped to gain greater insight into the potential role of timber management in maintaining avian diversity in the CNF. We hoped to answer three fundamental questions:

- 1) Is the quantity of habitat provided under the expected harvest regime sufficient to support viable populations of each species?
- 2) What is the impact of expected harvest levels upon available habitat for each bird species?
- 3) What is the impact of natural disturbance upon available habitat for each bird species?

For the first question, we estimated the quantity of habitat available for each species and estimated the number of breeding territories this could support. For the second question, quantities of available habitat for each species were compared among models with

expected levels of harvest, models without any harvest, and models of greater harvest intensity. For the third question quantities of available habitat for each species were compared between models containing only natural disturbance (no harvest), and models lacking any disturbance whatsoever.

STUDY AREA

Our study area was comprised of the Tellico, Hiwassee, and Ocoee Ranger Districts which make up the southern half of Cherokee National Forest (Figure 1*). These three districts cover 109,535 ha of forest, ranging in elevation from 231 m to 1530 m above sea level. Our study area is part of a complex of public lands in the Southern Blue Ridge physiographic province. It is bordered to the north by the Great Smoky Mountains National Park, to the east by the Nantahala National Forest (North Carolina), and to the south by the Chatahoochee National Forest (Georgia). The Tellico, Hiwassee, and Ocoee rivers, cross or originate in our study area. There are also numerous other watersheds throughout these districts.

Yearly average precipitation for the Citico Creek drainage across 11 years was 178 cm. Air temperatures varied from -25° C to 34°C with a mean temperature of 12.4° C. Average relative humidity was 86.1% (TVA 1968).

Forests on our study area were roughly subdivided into six broad forest types: Cove Hardwood, Hemlock/White Pine, Mixed Hardwood/Pine, Northern Hardwood, Oak/Hickory, and Yellow Pine. Forest types were classified based upon ≥50% dominance (by stem count of dominant and co-dominant species) by one tree species or

≥70% dominance of two to three tree species. A variety of factors including elevation, land form, soil type, moisture regime, and historic disturbance determine the distribution of forest types across Cherokee National Forest (USFS 1979, SAMAB 1996). Harvest regimes for a stand are largely determined by the forest type of that stand.

Principal Forest Types

Cove Hardwood- Cove hardwoods are dominated primarily by yellow poplar (see scientific names in Appendix C.), white oak, and/or northern red oak. Other tree species in this forest type include black oak, eastern hemlock, and black gum. A diversity of shrubs and forbs are usually present, including rhododendron, dog hobble, spice bush, and serviceberry. These sites represent the forest type of greatest botanical diversity and productivity. These sites are usually mesic. They are located on northerly slopes, coves, ravines, and moist flats. Cove hardwoods range in elevation from 150 - 1200 m. These sites have historically been managed for timber on a ninety-year rotation.

Hemlock/White Pine- Trees included in this community are eastern white pine, eastern hemlock, yellow birch, sugar maple, yellow poplar, basswood, northern red oak, and cucumber tree. Shrubs include wild hydrangea, dog hobble, rhododendron, and mountain laurel. These communities are typically impoverished in botanical diversity due to heavy canopy closure and thick needle layers. This forest type typically occurs above 500 m in moist, cool environments in coves and north-facing ravines. These sites have historically been managed on a ninety-year rotation.

* All tables and figures can be found at the end of this document.

Mixed Hardwood/Pine- These sites are co-dominated by hardwood and softwood trees. Dominant species included chestnut oak, southern red oak, pitch pine, shortleaf pine, table mountain pine, Virginia pine, white oak, and/or white pine. These sites are mesic to xeric, upland sites and/or steep southerly or westerly facing slopes. Understory species include blackgum, sourwood, sassafras, greenbrier, and *Vaccinium* spp.. These sites often have a history of fire or agriculture and are in transition from fire-adapted pine species to oak-hickory forest type. These stands are typically of mid to low productivity and are managed on a fifty to eighty-year rotation depending on the proportion of softwood to hardwood. They are typically found below 900 m in elevation.

Northern Hardwood- This forest type is dominated by sugar maple, yellow birch, and American beech. Secondary species include black birch, basswood, red maple, hemlock, northern red oak, and black cherry. This forest type has a diverse woody understory including striped maple, mountain maple, hobblebush, fire cherry, alternate-leaf dogwood, and silverbell. These communities are found above 1000 m on sites of high moisture and fertility. These sites have been proposed to be managed on a ninety-year rotation, though little harvest has been done since their acquisition, due primarily to slow growth, poor stem form, high visibility, and the presence of endangered northern flying squirrels.

Oak/Hickory- Predominant tree species in these mesic to xeric sites are southern red oak, white oak, post oak, black oak, chestnut oak, scarlet oak, pignut hickory, mockernut

hickory, red maple, shortleaf pine, Virginia pine and pitch pine. Shrub species include sourwood, redbud, Carolina buckthorn, black haw, farkleberry, and mountain laurel. These sites occur on ridge tops, mountain slopes, dry valleys, and a variety of other sites. This forest type is the most abundant and widespread in Cherokee National Forest. It is usually found below 900 m. It is managed on a forty-five to one-hundred-thirty-year rotation.

Yellow Pine- Yellow pine sites are dominated by pitch pine, shortleaf pine, table mountain pine, and/or Virginia pine. Associated tree species are the drier oaks, including white oak, blackjack oak, chestnut oak, and southern red oak as well as blackgum and hickories. This forest type occurs most frequently in old fields, infertile ridges, dry flats and slopes. It is typically found below 900 m. It has historically been managed for small pole timber and pulp on a forty to eighty-year rotation.

History of Timber Management

Timber harvest has gone through several phases since the establishment of the national forest (Figure 2). Upon acquisition little harvest occurred, because nearly all mature timber had already been removed, but also because the control of chestnut blight and wildfire took priority. From the mid 1960s until the early 1980s, attempts were made to diversify the age structure of the national forest. Average annual harvest from 1965 to 1985 was increased to roughly 750 ha, a five-fold increase over average annual harvest for the previous twenty years. In the late 1980s, harvest levels declined substantially. The average annual harvest since 1988 has steadily declined both in board feet and total

acres regenerated. This decrease can be attributed to several causes. Increasing environmental restrictions and bureaucracy surrounding proposed cuts on federal lands have caused a shift of harvests to private lands (SAMAB 1996).

Volume harvested as well as the intensity of harvest differs markedly among forest types. This is due in part to the relative abundance of the various forest types, but is also due to differences in the suitability of harvest, accessibility, and requirements unique to each forest type to ensure adequate regeneration (Table 1). Mixed pine-hardwood contains the largest acreage in the southern districts, followed closely by yellow pine and oak/hickory. Yellow pine clearly dominates in total acres harvested (Table 2). In addition, yellow pine and hemlock/white pine have had a much larger percent of their total acreage harvested than other forest types. Other forest types, especially the northern hardwoods, have had only limited harvests since the establishment of CNF. The timber management goals for CNF are to harvest high-quality sawtimber on a sustained yield basis. The pursuit of these goals combined with low levels of harvest will result in a continued increase in the average stand age and small quantities of early succession habitat.

METHODS

Bird habitat models constructed by Dr. Randy Dettmers, University of Tennessee for a related project were adapted to characterize bird habitat quantity and quality. Models were based upon five years (1992-1996) of point count data conducted by the author and John Bartlett (Bartlett 1995). Stepwise logistic regression (SAS 1990) of the point count data using the USFS CISC database as predictive habitat variables yielded

bird habitat models for twenty-five bird species. Six of these twenty-five species which performed well in cross validation with Georgia and Virginia data were included in our analysis: Acadian flycatcher, blue-headed vireo, chestnut-sided warbler, tufted titmouse, yellow-throated warbler, and yellow-billed cuckoo. These species were chosen to represent a range of elevation, forest type, successional stage, nest placement, and feeding guilds.

Bird habitat models used five predictive variables: forest type, condition class, stand age, site index, and elevation (Table 3). Elevation was not part of the CISC database and was added using Arc Info and SAMAB data (1996). The bird habitat models estimated the probability of occupancy for each of the six bird species for each stand in the CISC database. We multiplied the probability of occupancy of a stand by that stand's acreage to create a quantitative and qualitative measure of habitat for each bird species in each stand. We defined this measure as habitat units for a given species, much like habitat units for a Habitat Suitability Index model (Schroeder 1983). Summing these habitat units (hu) across the forest for each bird species yielded a measure of total habitat availability in CNF for each bird species. Dividing the summed habitat units for a species by the average breeding density of that species (Table 4) estimated the maximum number of breeding pairs that could be sustained in CNF. This measure assumed that 10 ha with a probability of 0.1 for a species was equal to 1 ha with a probability of 1.0 for that species. Estimates of the potential number of breeding pairs were compared to the MVP standard of 250 breeding pairs as recommended by PIF (C. Hunter, pers. comm.).

Concern that large numbers of low probability stands would inaccurately inflate my results prompted me to generate a second measure of habitat availability which

focused on high probability stands. Acreages of suitable habitat were derived in the same manner as habitat units described above, except that before summing these high probability habitat units, all stands with probabilities of occurrence below the probability associated with 75% specificity (the percentage of stands classified as occupied which are indeed occupied) were excluded. This cutoff probability where specificity equaled 75% varied by species and ranged between 0.14 and 0.30. Although some low probability stands were still included in this calculation, this reduced the number of stands deemed suitable by an average of 30% (S.E. = 10.94).

To evaluate the effects of alternative management plans upon habitat availability for these six bird species, the harvest activities of several management plans were simulated over the period from 1993 to 2053, and future CISC databases were created based upon the management alternatives. To do this, a SAS-based forest model was developed which simulated the timber harvest activities of management plans and updated stand records in CISC accordingly (SAS 1990)(Appendix A). The effects of even and uneven-aged harvests on stand conditions were simulated in selected stands deemed as suitable in the CISC database. Since harvest methods and intensities differ by forest type, the areas harvested, relative proportions of even-aged to uneven-aged harvest, group sizes, percents of stands harvested, ages of harvest, and condition classes harvested were specific to each forest type. These specifics were based upon past harvest practices and the expert opinions of the district silviculturists. Four types of natural disturbance were also simulated: fire, ice and wind, southern pine beetle, and hemlock wooly adelgid. The character and extent of natural disturbances were based upon existing literature and historic records (Appendix A).

During each simulation, CISC databases, including stand ages and condition classes, were updated as harvest and natural disturbance accumulated over time. For each simulation, new CISC databases containing these changes were created at ten-year intervals.

Four management plans were simulated, each with a different intensity of timber harvest: no timber harvest, expected harvest levels (based upon the expert opinion of district silviculturists), 200% of expected harvest intensity, and 300% of expected harvest intensity. An additional simulation was run with no timber harvests or natural disturbances to gauge importance of natural disturbance relative to timber-harvests. Habitat units were calculated for each simulation and summed for the six bird species in each simulation and graphed to compare habitat availability for each species under different management plans over time.

A sensitivity analysis was conducted upon the forest simulation variables to test the significance of these variables and their relative importance (Appendix B). The forest simulation was run repeatedly, systematically manipulating input variables at expected harvest levels, and 30% above, and 30% below expected harvest levels. To reduce the number of iterations necessary, the Optex procedure (SAS 1990) was used to identify a subset of variable settings in the sensitivity analysis. This greatly reduced the number of simulations needed to run the sensitivity analysis. The response of the total habitat units for each species to each variable was then tested using the general linear model procedure (GLM) (SAS 1990). Results from the sensitivity analysis quantified the importance of each variable independent of the relative abundance of each forest type. This facilitated comparisons of the influence of management on one acre of one forest type with

management on one acre of another forest type. Although the sensitivity analysis may have indicated that one forest type is more influential than another on a per acre basis, this relationship may not hold true at the forest wide level since some forest types are not as abundant as others. Also note the sensitivity analysis considered total habitat units, while analyses by forest type considered quality habitat units.

RESULTS

Species Accounts and Model Results

Acadian Flycatcher

Acadian flycatchers were detected at 31 of 215 points. Acadian flycatchers were found at low to mid-elevations in our study area. They generally occupied sites late in succession. The Acadian flycatcher model was based on positive associations with hemlock/white pine, cove, and northern hardwood forest types (the three most mesic forest types) and negative associations with oak/hickory and mixed forest types (dry forest types) (Table 5). In addition, a positive association with stand age and a negative association with elevation were included in the model. Max-rescaled R-square was 0.3688, correct classification was 85.0% (concordance), and the Hosmer-Lemeshow test indicated good fit of the model to the data ($P = 0.4509$).

At expected timber harvest levels, Acadian flycatchers were predicted to have an average of 44,403 habitat units (hu) available between 1993 and 2053 (Figure 3). Assuming an average breeding density of 14.5 pairs/40 ha (Hamel 1992), this could support a population of up to 16,096 breeding pairs, well above the estimated 690 hu needed to support a MVP of 250 pairs. Acadian flycatchers were projected to be the most

abundant of the six species. At expected harvest levels, this species would have a 164% increase in hu, the greatest increase of the six species. Increased harvest intensity slowed the growth of available habitat units. A 300% increase in harvest intensity resulted in 41% fewer hu than at expected harvest levels; however, even at this harvest intensity, the total number of hu increased 79% between 1993 and 2053. High-probability habitat units averaged 5% fewer than total habitat units (Figure 3). At expected harvest levels, high-probability habitat units averaged 42,752 hu between 1993 and 2053. There was little difference between high probability habitat units and total habitat units between 1993 and 2053 under any harvest scenario.

At expected harvest levels, mixed pine/hardwood ($\bar{x}$ = mean from 1993-2053) ($\bar{x}$ = 12,356 hu), yellow pine ($\bar{x}$ = 12,198 hu), cove hardwoods ($\bar{x}$ = 8,120 hu), and oak/hickory ($\bar{x}$ = 7,737 hu) provided the most high probability habitat. Habitat in these forest types increased steadily between 1993 and 2053. Hemlock/white pine ($\bar{x}$ = 2,003 hu) and northern hardwoods ($\bar{x}$ = 336 hu) provided relatively little high probability habitat but remained relatively constant between 1993 and 2053.

Based on the sensitivity analysis of forest simulation variables (Table 6), habitat availability for Acadian flycatchers were inversely related to most disturbance types. Habitat availability was influenced by ($P \leq 0.05$) the following model parameters (in descending importance): area of even-aged timber harvest in the hemlock/white pine forest type (-), area of even-aged harvest in oak-hickory (-), area of even-aged harvest in mixed pine/hardwood (-), area of even-aged harvest in yellow pine (-), area of even-aged harvest in cove hardwoods (-), area of disturbance by ice/wind damage (-), area harvested of uneven-aged harvest of oak/hickory (-), and hemlock woolly adelgid (-) (Table 6).

Southern pine beetle and wildfire variables were not related to habitat availability of Acadian flycatchers ($P \geq 0.05$).

Loss of mature forest, especially hemlock/white pine had the greatest negative relationship to habitat availability for Acadian flycatchers. This was entirely offset by maturation of the forest over time because of limited harvest.

Blue-headed Vireo

Blue-headed vireos were detected at 68 of 215 points. We generally found this species above 1050 m in sawtimber stands of hemlock/white pine, cove, and northern hardwood forest types. The blue-headed vireo model included positive associations with elevation, hemlock/white pine, cove, northern hardwood, and mixed forest types (Table 5). A positive association with stand age and a weak negative association with site index were also included. Max-rescaled R-square was 0.5559, correct classification was 88.2% (concordance), and the Hosmer-Lemeshow test indicated a very good fit of the model to the data ($P = 0.8779$).

The blue-headed vireo was predicted to average 22,622 hu between 1993 and 2053 at expected harvest levels (Figure 4). Assuming an average breeding density of 8.6 breeding pairs/40 ha (Hamel 1992), the southern districts of CNF were estimated to support an average maximum population of 4,864 breeding pairs between 1993 and 2053. We estimated 1,163 habitat units are necessary to support a MVP of 250 breeding pairs. At expected harvest intensities, habitat availability was projected to increase 30% between 1993 (18,150 hu) and 2013 (23,861 hu), then remain almost constant between 2013 and 2053. Increased harvest intensities reduced the projected habitat availability,

however habitat availability followed a similar increasing trend. At 300% the expected harvest intensity, habitat units peaked at year 2013 (21,597 hu), dropped slightly by year 2033 (20,000 hu), then increased slightly up to year 2053 (21,240 hu). High probability habitat units were 30% below total habitat units on average. At expected harvest levels, high probability habitat units averaged 15,878 hu between 1993 and 2053. There was little qualitative difference between high-probability habitat units and total habitat units under any harvest scenario.

At expected harvest levels, oak hickory ($\bar{x} = 6,973$ hu) provided the most high probability habitat units. Mixed pine hardwoods ($\bar{x} = 3,204$ hu), yellow pine ($\bar{x} = 2,243$ hu), cove hardwoods ($\bar{x} = 1,742$ hu), and northern hardwoods ($\bar{x} = 1,521$ hu) also provided substantial amounts of high probability habitat. Hemlock/white pine contributed very few high probability habitat units ($\bar{x} = 194$ hu).

Blue-headed vireo habitat availability was inversely related to most disturbances (Table 6). Variables which influenced ($P \leq 0.05$) habitat availability included (in order of descending importance): area of even-aged harvest in oak/hickory (-), disturbance by ice/wind (-), area of even-aged harvest in mixed pine/hardwoods (-), area of even-aged harvest in yellow pine (-), area of even-aged harvest in cove hardwoods (-), area of uneven-aged harvest in oak/hickory (-), area of uneven-aged harvest in hemlock/white pine (-), and the area of uneven-aged harvest in yellow pine (-) (Table 6). Southern pine beetle, fire, and the hemlock woolly adelgid were not related to habitat availability for the blue-headed vireo ($P \geq 0.05$).

Even-aged timber harvests of oak/hickory forest types had the strongest negative relationship with habitat availability. Forest maturation had the strongest positive relationship with habitat availability. Variable rates of change in habitat units over time were produced by a categorical relationship with stand age. Sharp increases in hu during the first two decades, followed by nearly constant levels over the remaining time span were caused by a cohort of stands harvested in the 1980's maturing into pole and sawtimber condition classes.

Chestnut-sided warbler

Chestnut-sided warblers were detected at 29 of 215 points. I generally found this species above 1000 m in oak/hickory, cove, and northern hardwood forest types. It was strongly associated with disturbance and was frequently found along roads, near small canopy gaps, and in seedling/sapling stands. The chestnut-sided warbler model included positive associations with elevation, seedling/sapling condition class, hemlock/white pine, oak/hickory, cove, and northern hardwood forest types, and site index (Table 5). Max-rescaled R-square was 0.6484, correct classification was 95.6% (concordance), and the Hosmer-Lemeshow test indicated a good fit of the model to the data ($P = 0.6616$).

The chestnut-sided warbler was predicted to average 2,317 hu between 1993 and 2053 at expected harvest levels (Figure 5), the lowest amount of habitat of the six species. Assuming an average breeding density of 11.9 breeding pairs/40 ha (Hamel 1992), CNF was estimated to support an average maximum population of 690 breeding pairs between 1993 and 2053. We estimated 840 habitat units were necessary to support a MVP of 250 breeding pairs. At expected harvest intensities, habitat availability remained relatively

constant between 2003 and 2053. The addition of 180 habitat units increased the total habitat units by 8% between 1993 (2,196 hu) and 2053 (2,376 hu). Increased harvest intensities increased the projected habitat availability. At 300% the expected harvest intensity, habitat units increased up to year 2033 (4,601 hu), then decreased from 2033 to 2053 (3,692 hu). High probability habitat units were 33% less than total habitat units on average. At expected harvest intensity there was an average of 1,558 high probability habitat units between 1993 and 2053. There was little qualitative difference between high probability and total habitat units at any harvest intensity. However the importance of the oak/hickory forest type and the resulting increases in habitat up to 2033 (see below) was greater for high probability habitat units.

At expected harvest intensity, northern hardwoods ($\bar{x}$ = 662 hu) and oak/hickory ($\bar{x}$ = 520 hu) were the greatest provider of high probability habitat units. Cove hardwoods ($\bar{x}$ = 118 hu) and mixed pine/hardwoods ($\bar{x}$ = 113 hu) provided less habitat. Yellow pine ($\bar{x}$ = 48 hu) and hemlock/white pine ($\bar{x}$ = 34 hu) contributed very few high probability habitat units.

Chestnut-sided warbler habitat availability was positively influenced by most disturbances (Table 6) ($P \leq 0.05$) including (in order of descending importance): area of even-aged harvest in oak/hickory (+), area of disturbance by ice/wind (+), area of even-aged harvest in cove hardwoods (+), area of uneven-aged harvest in oak/hickory (+), percent of each stand disturbed by ice/wind (+), area of even-aged harvest in mixed pine/hardwoods (+) (Table 6). Southern pine beetle, fire, and the hemlock woolly adelgid were not related to habitat availability for the chestnut-sided warbler ($P \geq 0.05$).

Forest maturation had the strongest negative relationship with habitat availability. Harvests rates exceeded sustainable levels in the oak/hickory forest type in the 300% harvest scenario, causing variable rates of change (fluctuations) in habitat units over time. Sharp increases in hu during the first three decades, followed by declines over the remaining time span were caused by a cohort of stands harvested from 1993 to 2033. As these stands matured into poletimber stands without further suitable stands being cut habitat availability declined for this species.

Tufted Titmouse

Tufted titmice were detected at 50 of 215 points. They ranged up to 1500 m, approaching the upper elevation limit in our study area, but were most common at lower elevations. Our model included positive associations with hemlock/white pine, oak/hickory, cove, and northern hardwood forest types. Negative categorical relationships with elevation, condition class, and site index were also included. This model performed well (Table 5). Max-rescaled R-square was 0.2508, correct classification was 76.7% (concordance), and the Hosmer-Lemeshow test indicated extraordinary fit of the model to the data ($P = 0.9997$).

Tufted titmice were predicted to average 49,655 hu between 1993 and 2005 at expected harvest levels (Figure 6). Assuming an average breeding density of 12.1 pairs/ha (Hamel 1992) the southern districts of CNF could support an average of up to 15,020 breeding pairs. We estimated that 826 hu were necessary to support a MVP of 250 breeding pairs. Our simulations predicted tufted titmouse habitat to be relatively stable, increasing by 1.7 percent at expected harvest levels from 49,014 hu in 1993 to

49,887 hu in 2053. Increased harvest intensity caused very slight declines in habitat availability. Harvest intensities 300% greater than expected caused a 3.8 percent decline in habitat availability from 1993 to 2053. High probability habitat units were 7% less than total habitat units on average. At expected harvest intensities, high probability habitat units averaged 46,146 hu between 1993 and 2053. There was no qualitative difference between high probability and total habitat units under any harvest intensity.

Oak/hickory ($\bar{x}$ = 11,492 hu), yellow pine ($\bar{x}$ = 11,209 hu), mixed pine/hardwoods ($\bar{x}$ = 10,410 hu), and cove hardwoods ($\bar{x}$ = 10,134 hu) contributed nearly equal high probability habitat units at expected harvest levels. Hemlock/white pine ($\bar{x}$ = 1,850 hu) and northern hardwoods ($\bar{x}$ = 1,051 hu) contributed relatively few high probability habitat units at expected harvest levels. These relationships remained relatively constant between 1993 and 2053.

Tufted titmouse habitat availability was influenced (Table 6) ($P \leq 0.05$) by (in order of descending importance): (-) area of even-aged harvest in hemlock/white pine, (-) area of even-aged harvest in oak/hickory, (-) area of even-aged harvest in mixed pine/hardwoods, (-) area of even-aged harvest in cove hardwoods, (-) area of even-aged harvest in yellow pine, (-) area damaged by ice/wind, and areas of uneven-aged harvested in (-) oak/hickory, (-) cove hardwoods, and (-) yellow pine (Table 6). Although hemlock/white pine contributed relatively few habitat units compared to more abundant forest types, harvests of hemlock/white pine caused relatively large changes in habitat units for that forest type. Southern pine beetle, hemlock woolly adelgid, and fire were not related to habitat availability for tufted titmice ($P \geq 0.05$).

Habitat units declined over time as seedling/sapling stands succeeded into pole timber stands, and as sawtimber stands were harvested. These declines were offset by increased habitat availability as poletimber stands succeeded into sawtimber stands, and poletimber stands were harvested or disturbed. Habitat units for tufted titmice had a net increase over time. Later stages of forest succession were not strongly related to habitat availability because poletimber and sawtimber stands were of nearly equal habitat value for tufted titmice, resulting in relatively constant amounts of habitat over time.

Yellow-billed Cuckoo

Yellow-billed cuckoo were detected at 40 of 215 points. Yellow-billed cuckoo were found up to 1050 m elevation but were most common at lower elevations. This species frequented moist deciduous forest and woodland thickets. Our model included positive associations with oak/hickory and mixed forest types and weak negative associations with elevation and stand age (Table 5). Max-rescaled R-square was 0.2492, correct classification was 78.3% (concordance), and the Hosmer-Lemeshow test indicated excellent fit of the model to the data ($P = 0.8464$).

Yellow-billed cuckoo were predicted to have an average of 24,277 hu between 1993 and 2053 at expected harvest levels (Figure 7). Assuming an average breeding density of 6.1 pairs/40 ha (Hamel 1992), the southern districts of CNF could support up to 3,702 breeding pairs. At expected harvest intensities, yellow-billed cuckoo habitat declined by 20 percent: from 27,260 hu in 1993 to 21,591 hu in 2053. Increased harvest intensity reduced this decline. At 300% the expected harvest intensity, habitat units declined by 5 percent. High probability habitat units were 33% less than total habitat

units on average. At expected harvest intensities, high probability habitat units averaged 16,201 hu between 1993 and 2053. High-probability habitat units had larger declines at lower harvest intensities than total habitat units.

Mixed pine/hardwoods ($\bar{x}$ = 6,950 hu) provided the most high probability habitat but also experienced the greatest decline in habitat. Oak-hickory ($\bar{x}$ = 5,013 hu), and yellow pine ($\bar{x}$ = 2,806 hu) also provided substantial amounts of habitat. These also declined strongly between 1993 and 2053. Hemlock/white pine ($\bar{x}$ = 1,020 hu), cove hardwoods ($\bar{x}$ = 409 hu), and northern hardwoods ($\bar{x}$ = 1 hu) provided little high probability habitat for yellow-billed cuckoo.

Habitat availability for yellow-billed cuckoo was increased by most disturbances. Variables which positively influenced ($P \leq 0.05$) habitat availability for yellow billed cuckoo were (Table 6) (in decreasing order of importance): (+) area of even-aged harvest in mixed pine hardwood, (+) area of even-aged harvest in oak/hickory, (+) area of even-aged harvest in hemlock/white pine, (+) area of even-aged harvest in yellow pine, (+) damage by ice/wind, (+) area of uneven-aged harvest in oak/hickory, (+) area of uneven-aged harvest in cove hardwoods, (+) area of uneven-aged harvest in yellow pine, (+) area of uneven-aged harvest in cove hardwoods, and (+) damage by hemlock woolly adelgid (Table 6). Southern pine beetle and fire were not related to habitat availability for yellow-billed cuckoo ($P \geq 0.05$).

Decreases in habitat availability were caused by maturation of forest over time. Even-aged harvests in mixed pine and oak/hickory were the strongest positive influences upon habitat availability for this species.

Yellow-throated Warbler

Yellow-throated warblers were detected at 29 of 215 sites. We found this species in low elevation cove sites with a white pine component. The yellow-throated warbler habitat model included positive associations with hemlock/white pine, cove, and mixed forest types, as well as a positive association with stand age. Negative associations with elevation (weak), oak-hickory forest type, and site index were also included (Table 5). Max-rescaled R-square was 0.2051, correct classification was 78.0% (concordance), and the Hosmer-Lemeshow test indicated excellent fit of the model to the data ($P = 0.9541$).

The yellow-throated warbler was predicted to average 29,156 hu at expected harvest levels between 1993 and 2053 (Figure 8). Assuming an average breeding density of 5.1 pairs/40 ha (Hamel 1992), the southern districts of CNF could support an average of up to 3,717 breeding pairs. An estimated 1,961 hu were necessary to provide habitat for a MVP of 250 breeding pairs. At expected harvest levels, yellow-throated warbler habitat more than doubled (119% increase): between 1993 (18,542 hu) and 2053 (40,637). Increased timber harvests slowed this trend. Habitat availability increased 59 percent between 1993 and 2053 at 300% the expected harvest intensity. High probability habitat units were 19% below total habitat units on average; averaging 24,515 hu between 1993 and 2053. Increased harvest intensity had a proportionally larger negative influence upon high probability habitat units than total habitat units.

Mixed pine hardwoods ($\bar{x} = 10,574$ hu), yellow pine ($\bar{x} = 6,901$ hu), oak-hickory ($\bar{x} = 3,034$ hu), and cove ($\bar{x} = 2,601$ hu) provided substantial high probability habitat. High probability habitat increased greatly in these forest types between 1993 and 2053.

Hemlock/white pine ($\bar{x} = 1,354$ hu) and northern hardwoods ($\bar{x} = 50$ hu) provided little high probability habitat.

Yellow-throated warbler habitat availability was negatively influenced by most disturbances. Variables related ($P \leq 0.05$) to habitat availability were (Table 6) (in decreasing order of importance): (-) area of even-aged harvest in mixed pine/hardwoods, (-) area of even-aged harvest in hemlock/white pine, (-) area of even-aged harvest in oak/hickory, (-) area of even-aged harvest in yellow pine, (-) area of even-aged harvest in cove hardwoods, (-) area damaged by ice/wind, and area of uneven-aged harvest in (-) oak/hickory, (-) yellow pine, and (-) cove hardwoods (Table 6). Southern pine beetle, wildfire, and the hemlock woolly adelgid were not related to habitat availability of yellow-throated warblers ($P \geq 0.05$).

Increasing forest age dominated habitat availability for this species. Harvest in mixed pine/hardwood, yellow pine, and oak/hickory had the strongest negative effects upon habitat availability of yellow-throated warblers.

DISCUSSION

Minimum Viable Populations

Estimated habitat availability for all harvest levels modeled exceeded our estimates of the minimum habitat units required to sustain a viable population of 250 breeding pairs as recommended by PIF (Figures 3-8). Estimated populations ranged from a maximum of 26,630 breeding pairs of Acadian flycatchers (year 2053, no-harvest

model, high probability habitat units) to 320 breeding pairs of chestnut-sided warblers (year 2053, no-harvest model, high probability habitat units) (Figures 3 and 5).

As with all forecasts, uncertainty increases as one ventures further into the future. Acadian flycatcher habitat may not exactly triple, nor will yellow-throated warbler habitat exactly double in quantity. What is certain is that late succession species will enjoy increases in habitat if timber harvests continue at reduced rates, at least for the next few decades. Habitat models of late succession species with categorical relationships to stand age or some derivative, (e.g. tufted titmouse and blue-headed vireo) displayed the most likely trend. Habitat for these species increased at expected harvest intensities, but at rates which levelled off in time as the forest reached equilibrium (Figure 3,4,6,8). Whether continuing to increase, or leveling off, habitat for these four species is likely to remain well above MVP levels for near future.

The remaining species, chestnut-sided warbler and yellow-billed cuckoo, suggest a different trend. While neither of these species were forecast to become extirpated, nor to even dip below projected MVP levels, they were forecast to experience strong declines in habitat availability at low harvest intensities (Figures 5,7). High probability habitat units for chestnut-sided warbler decreased to 320 breeding pairs in the no-harvest simulation, 70 pairs above the minimum viable population. However, at expected harvest intensities, breeding pairs increased slightly from 402 pairs in 1993 to 475 breeding pairs in 2053. For this species, timber harvest played a substantial role in maintaining habitat. Expected harvest rates may provide adequate habitat for this species, however, these estimates may be optimistic. Over half of the chestnut-sided warbler high probability habitat units at expected harvest levels were in mature stands with otherwise high quality

attributes. If these stands were deemed unsuitable because of advanced stand age, available habitat for this species may fall below MVP levels.

Yellow-billed cuckoo also showed substantial declines at all but the highest harvest intensity. Although this species was not considered an early succession species (Hamel 1992) our models included a continuous negative association with stand age. As a result, forecasted declines in yellow-billed cuckoo habitat were caused by forest maturation.

Bird species adapted to natural or artificial disturbance may be largely dependent upon human imposed management regimes to provide adequate habitat (Schulte and Niemi 1998). In the face of low intensity forest management with little natural disturbance, forest maturation may be a significant threat to many wildlife species, including the chestnut-sided warbler. Early succession species less common than chestnut-sided warblers may not have viable populations over the next fifty years. Additional models for early succession species should be developed to test this hypothesis. Chestnut-sided warblers were found in significantly lower abundance in burned sites than clearcuts (Schulte and Niemi 1998), and are even less abundant in mature stands (Hamel 1992). New England cottontails may also be seriously threatened by forest maturation in some areas (Litvaitis and Villafuerte 1996). Thomas (1990) stated that species occupying short lived habitat (i.e. early succession species) require larger minimum viable populations. Given that early succession species may have a declining habitat base at expected harvest intensities.

It might be argued that a MVP level of 250 breeding pairs is a low standard for common species and large parcels of land. Indeed these MVP levels are more commonly

used to gauge the effectiveness of small isolated preserves (measuring tens, hundreds, or thousands of hectares), small populations, or species with very large home ranges or sensitivity to people (such as grizzly bear) (Salwasser et al. 1986:159-173, Culotta 1995, Lande 1995). To hold CNF to similar standards may not be a reasonable approach to maximizing the effectiveness of our national forests in protecting wildlife diversity. Care must be taken when using MVP as a guideline to conservation. The application of a standard numerical value as a rule of thumb across a broad range of circumstances may be “misguided” (Ewens et al. 1987:59-68, Soule 1987:1-10). More demanding, size independent standards, based upon productivity per unit area, or breeding densities per unit area may be a better approach to evaluating land parcels which exceed these MVP levels (Van Horne 1983, Gibbs and Faaborg 1990, Trine 1998). Large parcels of public land often serve as population sources, maintaining species diversity across marginal or sink habitats throughout the region (Askins et al. 1987, Brawn and Robinson 1996). Simply maintaining populations above the MVP standard ignores this important potential role of national forests.

Expected vs. No-harvest Alternatives

There were few qualitative differences between the harvest intensities for most species. Each progressive increase in harvest added or detracted a proportional amount of habitat for each species. These accumulated into substantial differences in the quantity of habitat for each species in the expected harvest and no-harvest simulations. By the year 2053, the difference between no-harvest and expected harvest levels ranged from a decline of 2,154 hu (11% of no-harvest hu at 2053) for yellow-billed cuckoo to and

increase of 9,820 hu (13% of no-harvest hu at 2053) for Acadian flycatchers (Figures 3, 7). The accumulation of these differences should slow gradually as the forest asymptotically approaches a steady state. This appeared to be the case for the blue-headed vireo and tufted titmouse simulations (Figures 4, 6). For most species, habitat differences between expected and no-harvest simulations constituted a small proportion of their total habitat (2-13%).

However, there were substantial differences in chestnut-sided warbler habitat between expected and no-harvest simulations. A 23% decline in total habitat units in the no-harvest simulation was absent from expected harvest simulations. Expected harvests of oak/hickory, cove hardwoods, and mixed pine/hardwoods added roughly 600 total chestnut-sided warbler habitat units over the next fifty years, 30% of this species' available habitat. High probability habitat units at expected harvest intensities increased even more: roughly 45% greater than no-harvest simulations.

The importance of expected harvest levels varies by species. The large proportion of CNF deemed unsuitable for timber harvest provided a substantial buffer for late succession species. Even if all suitable stands in CNF were harvested, stands unsuitable for timber harvest would provide habitat exceeding MVP estimates for late succession species. In our simulations, abundant habitat for late succession species made the effects of expected timber harvests on these species relatively small. Less common species may not be as resilient. Early succession species, many of whom naturally occur at low frequencies, may be strongly influenced by timber harvests. For this to be the case, recently harvested stands must provide habitat suitable for these species. Our research suggests this may be the case.

Effects Of Natural Disturbance

The effects of the simulated natural disturbances can be quantified by comparing no-harvest simulations with the no-disturbance simulation. The no-disturbance simulation also measured the effects of time upon habitat availability since any changes in the quantity of habitat in this simulation were caused solely by time. Natural disturbance had a relatively small influence upon habitat availability in our simulations. There were few differences between no-cut and no-disturbance habitat simulations (Figures 3-8). The sensitivity analysis also suggested that natural disturbance played a minor role in our simulations (Table 6). Although ice/wind damage was significant for all six species, F values were relatively small for most species compared to F values for many of the harvests. However, the effects of ice/wind damage were relatively important for chestnut-sided warbler. The hemlock woolly adelgid was the only other form of natural disturbance which significantly affected bird habitat availability. It was significant for two species: yellow-billed cuckoo and Acadian flycatcher. Associated F values were extremely small, and almost non-significant. From this it is reasonable to conclude that natural disturbance played a very minor role in determining habitat availability for most bird species in our simulations.

Natural disturbance may significantly affect bird habitat in ways different from timber harvests. Our simulations were limited to variables in the CISC database, which contained no variables for disturbance type. Stands which experienced regeneration from timber harvest or natural disturbances had very similar CISC values. For example, simulated clearcuts and simulated wildfire stands had identical changes made to their

CISC variables: condition class and stand age were reset. However, numerous studies have detected significant differences in bird habitat between naturally disturbed sites and harvested sites. Schulte and Niemi (1998) found significant differences in bird species richness and abundance, and in several key habitat characteristics of logged and burned sites. Burned sites had higher densities of dead trees, wider size ranges of dead trees, greater shrub heterogeneity, and few living red maple. Differences have also been found in the foraging niches of two species of woodpeckers that are affected by burning (Murphy and Lehnhausen 1998). Black-backed woodpeckers specialized upon dying conifers 2-3 years after fires, while three-toed woodpeckers fed in less intensely burnt sites. Murphy and Lehnhausen (1998) concluded that the amount of fire across a landscape would likely alter the abundance and relative proportions of these two species. Closer to our study area, Newbold (1996) found significant differences in many vegetation characteristics between-tornado disturbed sites and clearcuts. He also found significant differences between these sites in avian communities, although measures of avian diversity were similar. Cavity, canopy, and ground nesting birds were more common in the tornado site than the clearcut, while shrub nesters were more common in the clearcut site. The differentiation of naturally-disturbed sites and harvest sites using our CISC variables within the confines of our bird habitat models was not possible. Although adequate data could be found to give rough estimates of the areas affected by natural disturbance, additional research is needed to develop accurate models of avian response to the diverse array of natural disturbances.

The importance of natural disturbance to some bird species may not be proportional to the magnitude or intensity of the disturbance. Disease fostered in ice or

fire damaged trees creates snags, providing nesting sites for cavity nesters such as the tufted titmouse, or foraging sites for woodpeckers. Increased shrub and canopy diversity from windthrow may substantially improve habitat for many species in our study area, including Acadian flycatcher (Bartlett 1995). The long term influences of natural disturbance also play a strong role in determining habitat availability. Most yellow pine and oak stands were established by disturbance decades ago. Natural disturbance likely plays a greater role in CNF than our simulations imply. Identifying and quantifying this role has proven elusive. Although there have been numerous silvicultural studies of natural disturbance, few wildlife habitat studies quantify its effects, and fewer modeling studies have attempted to include natural disturbance (Smith et al. 1981, Rottenberry 1984, Thompson 1993).

CONCLUSIONS

There is a general agreement among conservation biologists that the probability of survival for a population asymptotically approaches a maximum probability as population size increases: increases in population size yield diminishing returns (Belovsky 1987, Lande 1995). While the populations of the six simulated species appear viable in this study, the application of this theory to our simulations suggests that relatively small differences in habitat units between expected and no-harvest alternatives may yield large increases in the viability of disturbance dependent/early succession species, such as the chestnut-sided warbler. Expected harvest levels only marginally decreased habitat for late succession species and slightly increased habitat for forest age generalists. Harvest at these low levels probably had little effect upon the viability of these types of species with abundant habitat.

Intense timber harvests (200%, 300%) had significant effects upon the habitat availability of the six modeled bird species. The significance of timber harvests within each forest type varied across the six simulated species. Timber harvests, even at three times the expected intensity, did not jeopardize the viability of any species in our simulation.

Natural disturbance had little influence upon the habitat availability of the six species in our simulation, but may be more important in reality (e.g. Newbold 1996, Raebenold et al. 1998). In our simulations, natural disturbance by itself and in the absence of stand-replacing fires, may not provide adequate habitat to maintain all early succession species. Maturation of CNF due to harvest rates well below historic levels was the principal factor influencing bird habitat in our simulations.

Early succession species, especially uncommon early succession species, may be most in need of management. It may be possible to use forest simulations to develop management guidelines for early succession species. By identifying the eventual number of habitat units provided at specific harvest levels, one could estimate a minimum rate of harvest in each forest type necessary to provide habitat above a minimum viable population. First, intensive species specific studies must be undertaken to adequately sample the habitat relationships of the rare species. The accuracy of any simulation of a natural process is limited by the detail of the model, the range and quality of data available, the quality and detail of variables, and the range of forecast. Further research is needed to better quantify the effects of uneven-aged harvest and natural disturbance upon bird habitat.

PART II

**HABITAT CHARACTERISTICS AND NEST SUCCESS OF
GOLDEN-WINGED WARBLERS IN THE SOUTHERN BLUE
RIDGE**

The golden-winged warbler is uncommon throughout its breeding range, though it can be locally abundant where habitat requirements are met. Reductions of early succession habitat (Litivaitus 1993), regional declines in golden-winged warbler (GWWA) populations, and the disappearance of GWWA from much of its former range has raised concerns about its future. This species was placed upon the Blue List in 1981 (Tate and Tate 1986). Since 1986 it has been considered a "species of concern" by the United States Fish and Wildlife Service (USFWS) and the Audubon Society. It is listed as endangered in Ohio and Indiana (Hands et al. 1989) and threatened in Kentucky (Monroe 1994). It is ranked as a high priority species of concern in the Southern Blue Ridge (SBR) and Cumberland Plateau in the Partners in Flight (PIF) species prioritization scheme (Hunter et al. 1992, Gaines and Morris 1996). It is scheduled to be listed as "In Need of Management" by the Tennessee Natural Heritage Program (C. S. Major, pers. comm.).

GWWA nest from northern Wisconsin and Michigan into southern Ontario, and southern New England southward through Pennsylvania, Ohio, northern Illinois and Indiana, and along the Appalachians through West Virginia, Kentucky, Virginia, Tennessee, North Carolina, and into Georgia (Confer 1992a). This species winters in southern Central America and the West Indies (Figure 6). During the breeding season GWWA specialize on disturbed sites, often following timber harvest or in abandoned pasture (Confer 1992a). GWWA also inhabit several habitat types restricted to a part of its range including aspen clearcuts, alder bogs, tamarack bogs, regenerating strip mines, and wetlands (Burleigh and Hall 1958:501-502, Hands et al. 1989, Confer 1992a, Confer 1992b, Canterbury 1993a, Bucklew 1994:138). GWWA have also been reported to use

powerline rights of way and roadsides throughout much of their range (e.g., Burleigh 1958:501-502).

GWWA nest from May to June. Nests are placed on or near the ground. The base of the nest is comprised of leaves. The upper bulk of the nest is built with loosely woven, broad-leaved grasses. The cup is lined with bark shreds, fine grasses or hair. Nests are usually placed in a clump of herbaceous vegetation near the edge of an opening in surrounding brushy habitat (Will 1986, Curson et al. 1994). Four to six eggs are usually laid (Curson et al. 1994). Incubation takes approximately ten to twelve days, and fledging another eight or nine (Will 1986). In the two published productivity studies of GWWA, Will (1986) found a hatching success of 74.8 percent, while Canterbury (1993) found nest success ranged between 57 and 86 percent. Will's (1986) study found productivity averaged 3.4 young per nest across the three years of study ($n = 28$). Nest failure in this study was largely attributed to predation by birds, snakes, or rodents because most 'predated' nests appeared undisturbed. Approximately 30% of GWWA nests were parasitized by brown-headed cowbirds in both studies.

Over the last century GWWA have shown significant population changes throughout parts of their range in response to human activities (Gill 1980, Sauer et al. 1997). Abandonment of farms across the Northeast in the early nineteenth century led to an apparent population boom of GWWA and other early succession species. This boom has been followed by a decline as agriculture related habitats succeed into mature forest (Gill 1980, Litvaitis 1993). Breeding Bird Survey data have shown national declines in detection of 3.3 percent annually for GWWA from 1966 to 1979 ($P = 0.01$) and increases of 3.2 percent from 1980-1996 ($P = 0.02$) (Sauer et al. 1997). The interpretation of these

results has been complicated by regions of range expansion into New York, New England, Ontario, Quebec, and Minnesota masking declines in this species from the southern and central parts of its range. GWWA have received increased attention lately because of their disappearance from many parts of their range, low numbers throughout their range, and the apparent decreases in available breeding and wintering habitat.

Many studies have investigated the relationships between GWWA and the closely related blue-winged warbler. Blue-winged warblers (BWWA) utilize similar habitat and may compete with GWWA (Will 1986). In addition, hybridization between these two species is common (Short 1963). GWWA populations seem to decline coincident with the arrival of BWWA to an area (Gill 1980). Some researchers hypothesize GWWA declines result from interactions with BWWA. Speculation as to the mechanism of competition include theories of genetic swamping, social dominance, and competition (Will 1986), although no firm conclusions have been reached.

Habitat loss is a possible alternative explanation of observed GWWA declines. Confer et al. (1981) studied habitat relationships in New York and found significant differences in a sample of eight GWWA and 16 BWWA territories, with BWWA using sites in later stages of succession. However, in a survey of historic records of that area, Confer rejected habitat changes as the sole cause of historic population fluctuations. In a later study (Frech and Confer 1987), four GWWA territories sympatric with BWWA territories were compared with two territories occupied by only GWWA. Significant differences in GWWA habitat were found, but the authors concluded that the differences resulted from inherent differences in the site and were not a consequence of displacement of GWWA by BWWA. Both of these studies were too limited in sample size to allow

any firm conclusions. In central Michigan, Will (1986) monitored the nest sites of 15 GWWA and 16 BWWA and characterized nearly one hundred habitat variables. Due to a lack of statistical power, significant differences were found for only five variables; GWWA had higher herbaceous diversity, and lower mean and summed DBH of sampled oaks and hickories. Will (1986) concluded that replacement of GWWA by BWWA was not driven by greater habitat specificity of GWWA.

In the Southern Blue Ridge (SBR), BWWA are uncommon, perhaps more so than GWWA. In addition BWWA and GWWA seem to be allopatric in the Southern Appalachians; BWWA are typically found below 600 m while GWWA are found above 600 m (LeGrand 1975, Hamel 1992). Because of this stratification by elevation, competition between these two species may be unlikely within this region.

In the Southeast, GWWA are so uncommon that they are rarely mentioned in the literature. There are few published studies of GWWA in Virginia, Tennessee, North Carolina, or Georgia, the southern extent of their range. Because the sighting of a GWWA is noteworthy, there are numerous historical records. William Brewster (1885) encountered GWWA at several locations near Franklin, North Carolina. Although Brewster explored elevations up to 1600 m, he only mentions GWWA below 1000 m. They were “generally distributed” in association with “second growth forests of oak, etc” (Brewster 1885). There are few other notes in the literature until after the extensive logging of these forests after 1900. This absence of records reflect inaccessibility of this region as opposed to the absence of this species. It is likely that widespread logging early in this century created abundant habitat; in these operations, no attempts were made to establish regeneration before or after cuts, and wildfire was a frequent problem (U.S.

Forest Service 1986). Following this era, GWWA are mentioned by numerous authors (Odum 1950, Conner and Adkisson 1975, Kendeigh and Faurer 1981, Wilcove 1988) in conjunction with forest edge or young clearcuts.

GWWA may be declining in the SBR in recent years. Locations historically known for their presence have been unoccupied in recent years (e.g. Stupka 1963). GWWA now appear to be extirpated from the Great Smoky Mountain National Park (Nicholson 1997, F. Alsop, pers. commun.), and nearly extirpated from Cherokee National Forest (N. Klaus unpublished data).

Innovative analyses of BBS data by James et al. (1996) supports this apparent decline. Two regions, the Blue Ridge Mountains and the Cumberland Plateau, show significant avian declines in both analyses of these data. Interestingly, of the six species of neotropical migrants with demonstrable declines, five are early succession species. GWWA is ranked as having the fifth largest declines. Because of low population levels in the region, these recent declines are not statistically significant ($P > 0.05$).

Today little is known of GWWA distribution and abundance within the SBR. No monitoring programs have focused on early succession habitat with adequate sample size. Detectability of GWWA is significantly lower than most other bird species. GWWA frequent dense habitat where visibility is low and travel is difficult. In addition many GWWA songs do not carry well and are in the upper limits of many observers' hearing range. Most problematic in the detection of GWWA is a sharp decrease in singing after acquiring a mate (Ficken and Ficken 1967). Most GWWA mate shortly after arriving on territory in early May, often before traditional point count dates and times, such that estimates of abundance based upon song counts underestimate population levels

(Canterbury 1994). As a result, little more than a handful of observations constitutes the extent of the Natural Heritage Database for GWWA in North Carolina, and Tennessee (H. Legrand, pers. comm.).

Plant and animal communities dependent upon disturbance have decreased in numbers or been relegated to other habitat types that may be lower in quality (Litvaitis 1993). It is likely that habitat for these species, including the GWWA, has long been provided by natural and aboriginal wildfire, ice and windstorms, beaver activity, insect pests and other disturbances in the SBR (Hunter *in prep.*). These disturbances have long been a component of the SBR ecosystem (SAMAB 1996, Delcourt and Delcourt 1997, Buckner and Turill 1998). Today natural disturbance is largely absent from the system resulting in a radically different flora than that of the previous 10,000 years (Harmon 1982, Buckner and Hamel 1998b). Fire, southern pine beetle and other disturbance is actively suppressed by state and federal land managers in the region. Widespread logging through the early 1900s, followed by relatively little logging since, has created an even-aged forest with an average age of 70 years. This has resulted in a forest with an unnaturally homogenous age structure that is unnaturally vigorous; disease is less prevalent, and elements associated with disturbance such as large downed woody vegetation and canopy gaps exist at low levels. It is possible that habitat for GWWA was created and maintained by wildfire set by aboriginals (Delcourt and Delcourt 1997, Buckner and Turill 1998), although little information is available to test this hypothesis.

It is unknown how much GWWA habitat in the SBR is publicly owned. The combined acreage of the Cherokee, Pisgah, Nantahala, Chattahoochee, George Washington and Jefferson National Forests, as well as the Great Smoky Mountain

National Park (1,280,408 ha) may account for nearly all of the GWWA habitat; private ownership is generally restricted to the lower elevations in the SBR (SAMAB 1996). Hence management decisions by the National Park Service (NPS) and the US Forest Service (USFS) may determine the fate of southern populations of GWWA. Changes in the quantity and method of logging on national forests of the SBR are underway which may impact regional GWWA populations. Decreases in total area harvested and longer rotations may decrease early succession habitat. Increased use of shelterwoods and uneven-aged management may diminish the suitability of these habitat for early successional species. Management aimed at reducing visual impacts by making cuts smaller, linear, and at lower elevations also has the potential to reduce the amount of habitat available to GWWA.

At issue is the decision by the USFS to play either an active or passive role in the management of national forests. Specifically, what role if any, should timber harvests and habitat manipulation play in maintaining populations of early succession species in the SBR? As a small part of the overall picture, considerable debate has erupted around the GWWA. Is quality breeding habitat provided by timber harvest? Are there sources of natural disturbance which will provide habitat for the GWWA in the SBR? What are the characteristics of good GWWA habitat in the SBR? The very legitimacy of this and other species' presence in the SBR fauna has been brought into question. While such fundamental questions are beyond the scope this study, this research attempts to take the first steps toward answering these questions. The objectives of the study were:

- 1) to identify key habitat parameters which define golden-winged territories and nesting sites in managed forests; and

2) to document the nest success and productivity of golden-winged warblers in managed forests.

STUDY AREA

Our study area was the southern districts of the Cherokee National Forest (Tellico, Hiwassee, and Ocoee districts), Tennessee and the Cheoah and Wayah districts of the Nantahala National Forest, North Carolina. These forests are located in the mountainous Southern Blue Ridge physiographic province. Elevation ranges from 730 to 1833 m. Soils are dystrochrepts, hanhapludults, and hapludults with mixed kaolinitic and micaeous mineralogy with mesic and udic moisture regimes respectively (SAMAB 1996). Temperatures and precipitation vary widely by season and elevation. Average annual precipitation is between 106 and 133 cm and is well distributed throughout the year. Precipitation exceeds evapotranspiration. Mean annual air temperatures range from 10° to 15° C but vary widely by season. Summer lows and highs are 10° and 32° C, respectively (Baldwin 1973).

Dominant vegetation is evergreen and deciduous forest. Variability in climate, slope, aspect, and hydrology has created diverse vegetation types (see Part I for an overview). This study focused primarily upon the Oak/Hickory, Cove, and Northern Hardwood Forest types.

Active timber harvests continue within the study area. In the last ten years, >90% of harvest operations have been even-aged, either clearcut or shelterwood. Harvest units have averaged 13 ha (USFS unpubl. data).

METHODS

Stand Selection

In 1997, we surveyed all forest stands of northern hardwood, cove hardwood, and mesic oak/hickory forest types that were <20 years old and above 1000 m elevation in the southern districts of Cherokee National Forests. Few GWWA were encountered and these were restricted to the northern hardwood and cove hardwood sites. To afford a greater sample size, we expanded our monitoring in 1998 to include sites in the Cheoah and Wayah districts of the Nantahala National Forest, North Carolina.

In 1998 in North Carolina, we randomly selected twenty stands in the Cheoah district classified in the USFS CISC database as cove, northern hardwood or mesic oak hickory forest type, less than twenty years old, irrespective of elevation. Although we found no birds in oak/hickory in 1997, several sources suggest it as possible habitat (Odum 1950, Hamel 1992). To increase sample sizes, we also sampled other stands with similar characteristics to the selected stand in that compartment.

In Tennessee, harvest was greatly reduced and few stands met selection criteria. We sampled all stands of cove, northern hardwood, and mesic oak hickory less than twenty years old and above 1000 m elevation (19 stands).

Survey Methods

All GWWA surveys were completed by the author between sunrise and 10 am. Listening and playbacks were used to detect singing males. The duration of the surveys was variable to allow for longer sampling times in larger stands; however a minimum of

10 min was spent in each stand. Because singing GWWA often stopped singing in response to playbacks, upon entering a stand I listened for a minimum of 3 min without using a playback, often longer for larger stands. During this time I moved to several locations in the stand to ensure that I could hear any singing male within the stand. If no birds were encountered during the initial listening phase, I began playbacks. I alternated the GWWA type I and type II songs (Ficken and Ficken 1968), playing 3 min of one song before changing to the other. Playback was conducted from several sites throughout the stands. If no GWWA were detected, I switched off the playbacks and listened for 3 min more. Several times I detected birds which had not been singing before playback and which apparently only responded to playback after it had ceased.

In stands with multiple birds, tracking individuals was difficult. Stands with multiple territories were those where two or more birds were located simultaneously on two or more occasions within distinctive territories or where multiple nests were found. This approach represents a conservative estimate and may underestimate some colony sizes. New territories continued to be found through early June after most nests had fledged young. By late June, birds were difficult to detect and responded poorly to playback even on known territories. Only one bird was heard after July 1st, and that was on a known territory. Birds were not detected with or without playback on known territories after July 1st.

Nest Searching

Six full-time nest searchers were employed from 10 May to 10 June, 1998. In addition, I served as a part time nest searcher during afternoons or when not surveying

new stands for occupancy. Each nest searcher was trained for several days, and accompanied more experienced searchers for several days following that before working independently. Nest searching technique was fairly simple. One to two individuals arrived at stand(s) known to be occupied. Searchers followed singing males to get rough ideas of the territory and to locate the female. If no female was located after several hours, nest searchers attempted to flush incubating females by carefully walking through likely looking areas (areas frequented by the male). This often resulted in one of three outcomes: 1) a female was flushed and the nest found; 2) one or both birds became agitated as searchers approached the nest, resulting in the nest being found; 3) no female was ever located and the territory was left to be re-searched at a later date. Very few nests were not found after a female had been detected.

To track changes in the mating status of territorial males, data on behavior and song characteristics were taken by nest searchers every day a territory was entered. Song frequency, duration, song type, and number of low notes in the type I song were noted. The presence or absence of a female was also noted. The regular presence of a female in a territory was believed to strongly suggest the male of that territory was mated. In territories where we had not sighted a female we estimated the mating status of the male by his singing behavior. We assumed that unmated males sang more frequently, were more likely to sing throughout the day, had a type one song with three or more bottom notes (Ficken and Ficken 1967). Territories with males believed to be mated were searched for a longer time and were more likely to be revisited at a later date. Since most territories were in clusters, changes in the behavior of males previously believed to be unmated were usually detected. All territories were searched two or more days.

Nest Monitoring

Upon finding a nest I placed flagging 5-10 m away, with a unique code to identify the nest, a compass bearing and distance to the nest. The status of the nest was checked and the number of eggs or young recorded. Approximate age of the young was also recorded. If the female was incubating or brooding, it was also noted. Nests were re-checked by one person every 4 d with the same data recorded until the fate of the nest was determined. Every attempt was made to minimize the time spent at the nest to avoid attracting predators. Nests were approached from different directions each visit to avoid making a trail. Where vegetation allowed, nests were checked with a mirror affixed to a 6 m pole to avoid leading predators to the nest.

Estimating predation rates was complicated by two factors: nest predators such as small rodents and snakes often leave no indication of their presence (Ettel 1998); and premature fledging in this species is common beyond 6 d after hatching (Will 1986). To accommodate these problems, we avoided checking nests beyond 6 d after hatching, but made every effort to check nests on day five or six. Some nests were checked after day six if weather made roads impassable leading up to day six. Nests found empty before 6 d after hatching were assumed to be predated. In calculating nest success, predation rates were assumed to be constant throughout the nestling stage. I extrapolated daily nest survival rates from day 6 to day 8 to determine nest survival for the entire nesting period, assuming the nestling stage lasted eight days.

Nest fate was deemed either successful or unsuccessful. For a nest to be deemed unsuccessful, one of several criteria had to be met: the nest was found empty and the

nestling were less than six days old; the nest was upset, turned on its side, or ripped apart (at any nest age); nestlings were found dead and no survivors were found in or around the nest. All other nests were deemed successful.

Daily survival rate for eggs and nestlings, as well as the overall nest success rate was calculated using the Mayfield (1975) method assuming a ten day incubation period and an eight day nestling period. The average number of young fledged was calculated for successful nests and all nests, and confidence intervals were calculated for these values.

Habitat Measurements

To model GWWA habitat, site-specific vegetation data were collected in July and August after nesting was complete (Table 7). To characterize GWWA breeding territories, we collected vegetation data from occupied stands centered on a frequently used song perch. We characterized a sample of occupied stands from throughout our study area. We did not characterize multiple territories of the same stand. The song perch identified for plot center was the most frequently used perch within the territory of a male where the male repeatedly sang when not involved in aggressive behavior with other birds or pursuit of a female. I assumed plots centered upon favored song perches represented breeding habitat. Surveyed stands deemed unoccupied were also sampled and plot centers randomly chosen so that the entire plot fell within the stand.

Vegetation was also characterized around known nests. One plot was centered on the nest and four additional plots were centered at 10 m from the nest in the cardinal directions (Figure 10). The same vegetation parameters were recorded for nests,

territories, and unoccupied sites. Vegetation was roughly subdivided into the following classes:

- **Trees and Snags** - woody plants ≥ 10 cm DBH and had the potential to reach the canopy. Most trees originated from a single stem. (For example: sourwood, linden). For trees we estimated basal area using a $(2.5 \text{ m}^2/\text{ha})$ factor prism. Every 'in' tree was identified to species, and the DBH and height were estimated, using a diameter tape to measure DBH and a clinometer and tape measure to measure height. The canopy cover was measured in the four cardinal directions, 5 m from point center using a spherical densiometer.
- **Saplings**- live trees < 10 cm DBH and > 1 m in height. All saplings ≥ 5 m from point center were identified to species, and an average height for each species was recorded.
- **Seedlings**- living trees < 1 m tall. All seedlings ≥ 5 m from point center were identified to species, and counted.
- **Shrubs** - woody plants which did not have the potential to exceed 10 cm DBH and 8 m in height. Shrubs often had multiple stems. (For example: farkleberry, rhododendron, blueberry). All shrubs ≥ 5 m from point center were identified to species, and counted.
- **Vines** - climbing plants either woody or non-woody (i.e. grape, crossvine, Virginia creeper). All vines ≥ 5 m from point center were identified to species, counted, and an average height estimated for each species.
- **Herbaceous plants** - all non-climbing, living, non-woody plants. The average height, ratio of grasses to forbs, and density category (1-5, 5 most dense) were visually

estimated. The three dominant species (species with the greatest percent cover) were identified to species, and their percent cover visually estimated.

Characteristics of nearby roads were recorded. For our purposes, roads were defined as any linear break in vegetation and/or slope caused by past operation of vehicles (including skidders). Roads were classed as:

- type 1 = paved road
- type 2 = gravel road with little or no vegetation growing in it
- type 3 = two-track gravel road with vegetation growing between the tracks
- type 4 = two-track gravel road mostly overgrown with vegetation such as grass and shrubs/small trees
- type 5 = not drivable, overgrown with small trees etc. (i.e. logging roads, skid trails)

Distance to the nearest road from plot center was measured using a laser range finder or tape measure. We also recorded slope and aspect for every plot. The elevation of every plot was estimated from USGS topographical maps. Stand age from the CISC database was also included as a habitat variable for each plot.

Variables were tested for significant correlation. Uncorrelated variables were included in stepwise logistic regression to test for differences between occupied and unoccupied sites, nest sites and surrounding points, nest sites and occupied territory sites, and nest sites and unoccupied sites (SAS 1990).

RESULTS

Playback Surveys

We detected GWWA in five of the nineteen stands surveyed in 1997 on the Tellico District of Cherokee National Forest. Elevation of occupied stands ranged from 1200 to 1460 m. Four of the five stands were surveyed in previous years in a related study and had been occupied by GWWA at least one or more previous years. Four occupied stands were northern hardwood forest, the fifth was cove hardwood forest. The average stand age of the occupied stands detected in 1997 was 10.6 years ($SE = 1.978$). No females or nests were detected, although in 1995 a female was seen carrying food at one of these sites (K. Calhoon, pers comm.). Point counts in the Tellico District of Cherokee National Forest from 1992 to 1996 detected two to three GWWA in 1992, 1993, 1994, and 1997, and one in 1995. No GWWA were detected during these point counts in 1998.

In 1998, 77 territories were found either in the randomly selected stands on the Nantahala National Forest or en route to these stands (i.e. part of the same complex of harvested stands as the one we were surveying) (Table 8). Most birds found were part of a loose community consisting of 2 to 31 territories across one or a series of stands. All birds found were on the Cheoah District, with the exception of the Big Choga group, which is on the Wayah Ranger District, and one bird which was found on the Tellico District.

Habitat Analysis

Breeding Territories

I used data from 26 unoccupied stands and 31 occupied stands in the analysis. Basal area was lower in occupied stands than basal area of unoccupied stands (Figure 11) (Table 9) ($P = 0.0310$). Roadtype was higher ($P = 0.0332$) in occupied stands than road type in unoccupied stands. Sapling density appeared to be greater ($P = 0.0557$) in occupied stands than sapling density in unoccupied stands. Average tree height also appeared to be greater ($P = 0.0655$) in occupied stands than tree height in unoccupied stands. These eleven variables accounted for 70 % of the variance (Max-rescaled R-squared = 0.7037) and had 93.5% concordance with the data. The Hosmer-Lemeshow test indicated an excellent fit of the model to the data ($P = 0.9959$).

Nest Sites

Twenty-three GWWA nests were found; two in 1997 and twenty-one in 1998. No nests were found in any of the randomly selected stands; however, most nests were found in neighboring stands of similar age and vegetation structure. One nest was placed approximately 1 m off the ground in thick multiflora rose brush; all others were placed on the ground in herbaceous vegetation.

Twenty-two nests were compared with the eighty-eight vegetation plots surrounding each nest. One nest site could not be included in our analysis because the site was mowed before we began vegetation sampling. Eight of the eleven uncorrelated

habitat variables used in the breeding territory analysis were included in this analysis (the remaining three did not differ between samples within the same stand i.e. elevation).

Herbaceous density was greater at nest sites than surrounding points ($P = 0.0097$) (Figure 12, Table 10). No other habitat variables differed among nest plots and adjacent plots ($P > 0.05$).

Twenty-two nest sites were compared with 31 territories/song perches. The number of sapling stems was greater at nest sites than song perches ($P = 0.0228$) (Table 11). Average canopy cover was lower at nest sites than at song perches ($P = 0.0327$) (Figure 13). The variability of canopy cover appeared to be greater at nest sites than song perches ($P = 0.0687$).

The twenty-two nest sites were also compared with the 26 unoccupied sites. Variation in canopy cover was greater at nest sites than at unoccupied sites ($P = 0.0075$) (Table 12) (Figure 14).

Nest Productivity

Mean clutch size was 4.5 (S.E. = 0.51). Of 23 nests found, 17 were successful (74%). Of the six unsuccessful nests, two were abandoned before eggs were laid, one was run over during incubation by a four-wheeler, and three were predated during the nestling stage. In two nests, some eggs or young were lost; in one three of four eggs did not hatch but remained in the nest until after the young fledged; in the other, heavy rains flattened the crown vetch surrounding the nest causing it to tilt. One of five young fell out and died of exposure. Mayfield (1975) statistics were calculated for all 1998 nests. Daily survival rate was 99% for eggs, and 98% for nestlings. Incubation nest success rate

was 90%, nestling nest success rate was 81%, and overall nest success rate was 72.5%. An average of 4.5 +/- 1.01 young were fledged per successful nest; an average of 3.7 +/- 1.92 young were fledged from all nests. Fledging generally occurred the first and second week in June. One nest did not fledge until June 23rd. This was likely a re-nest since it was not detected until June 3rd (just being built), five days after a nearby nest was found depredated. No GWWA nests were parasitized by brown-headed cowbirds although cowbirds were seen frequently at the Stecoah Gap site.

DISCUSSION

Habitat Requirements

Although I found GWWA in a variety of habitat types, I observed several components common to most GWWA territories: thick brushy habitat juxtaposed with patches of relatively open, herbaceous vegetation containing scattered small woody plants. Generally these formed a complex of GWWA territories which formed the loose colonies described in many GWWA studies (e.g. Confer and Knapp 1981).

Descriptions of GWWA habitat throughout their range mention a combination of herbaceous and woody patches. Ficken and Ficken (1968) described GWWA habitat in New York and Maryland as overgrown fields with many shrubs and small trees under twenty feet, bordered by taller deciduous trees. Nest sites were situated at field-woodland edges. Canterbury (1990) described occupied sites in West Virginia as oak/hickory forests interspersed with fields dominated by goldenrod, blackberry, and multiflora rose; white pine, black locust, and other trees interspersed with brushy thickets; and pine thickets interspersed with wetlands. Sites in Michigan (Will 1986) were a mosaic of

forest interspersed with abandoned agricultural fields and small scale gravel mining. Territories and nests sites were focused along edges between forest and openings. In Minnesota (Murray and Gill 1976), GWWA occupy tamarack swamps crisscrossed by man-made openings.

These features generally require some form of recent disturbance. Timber harvests served this purpose in our study area. Stand age was critical in determining habitat suitability. GWWA were only encountered in stands under twenty years of age in northern hardwood and cove sites during seven years of collecting point count data across all ages and forest types in the Cherokee National Forest,. During the 1998 field season, median stand age of occupied sites was 6 yr, unoccupied sites 12 yr. Minimum stand age of occupied sites was 3 yr. Stand age was lower in occupied stands than in unoccupied stands ($P \leq 0.05$) based upon a Wilcoxon-Rank Sum test. The decline in GWWA in CNF was also probably a function of stand age: the average age of the nineteen surveyed stands was twelve years, while ninety-five percent of all occupied stands (study wide) were less than eleven years old. Canterbury (1990), Confer (1981) and others have noted a decline in habitat quality as sites age. However, there is more to GWWA habitat than stand age.

No GWWA nests occurred in the naturally regenerating portion of the stand but were in openings adjacent to the harvested areas. These grassy areas adjacent to the seedling/sapling stands have had succession retarded through a combination of seeding with herbaceous cover plants and annual or biannual mowing. The openings came in many forms: wildlife openings, roads, powerline rights-of-way, or low areas where saturated soil retarded woody growth. The disappearance of herbaceous cover may cause

GWWA to abandon habitat otherwise suitable for breeding. Most unoccupied stands we surveyed had ample seedling/sapling habitat but lacked roads, log landings, or wildlife openings. These openings account for at least three of the differences between nest sites and song perches; nest sites had lower canopy cover ($P = 0.0327$), sapling density ($P = 0.0228$), and higher variability of canopy cover ($P = 0.0687$) (Table 11).

Clearly timber harvests have played an important role in providing habitat for GWWA in the SBR. Young stands with associated log landings and roads provide productive nesting and foraging habitat for GWWA in the Nantahala National Forest.

Productivity

Mean clutch size and mean number of young fledged were within the ranges observed by Will (1986) and Canterbury (1993b). These studies did not use the Mayfield method so direct comparisons are difficult. Totaled across three years of study, Will (1986) found eleven of thirteen (85%) GWWA nests successfully fledged at least one young excluding re-nests, nests abandoned before incubation and nests with incomplete data. Applying this standard to our data, fourteen of eighteen nests fledged at least one young: a 77% success rate. Data reported by Canterbury (1990, 1993b) are even more difficult to compare. Assuming that data reported in his 1993b paper included all nests found (i.e. abandoned nests, renests, etc.), then nest success averaged 68% for 28 nests over five years. By this standard, our nest success averaged 83% for all 1997 and 1998 nests. In spite of the small sample size that plagued all of these studies, it seems likely that our measure of nest success is comparable to that observed by Will and Canterbury. A seven year nesting study in New York by Confer has the best sample sizes of any study

to date (J. Confer, pers. comm.). Across 21 sites comprised mainly of abandoned farmland, 27 of 57 (47%) GWWA nests (including renests) which had at least one GWWA egg, but were not parasitized by brown-headed cowbirds, fledged at least one young. Seven of twenty-nine (24%) parasitized nests with at least one GWWA egg (including re-nests) fledged at least one GWWA young. Combined, this yields a 39% nest success. There is little doubt that nest success at these sites was significantly different from our sites, in part because of cowbird parasitism.

Although cowbirds were occasionally observed at the Stecoah Gap site, no cowbird eggs or young were found in GWWA nests (including two nests at the Stecoah Gap site). This seems surprising since the Nantahala National Forests contains numerous inholdings. The four sites with the most GWWA were found on ridges with private pasture less than 1 km away. Cowbirds have been shown to parasitize GWWA, occasionally at high levels (Will 1986, Coker and Confer 1990) but may not significantly reduce GWWA productivity in the SBR. Cowbirds in CNF generally do not fly into forested areas to parasitize birds but stay near agricultural edges (Buehler unpublished data). Because of this, harvested stands may provide higher quality habitat than the classic abandoned field/field edge habitat.

Clearly, GWWA occupying seedling/sapling stands were highly productive in 1998. Since reproductive success can vary widely by year, further study is needed to confirm the benefits of timber harvest to GWWA in the SBR.

CONCLUSIONS

Like most wildlife dependent on early succession habitat, GWWA face an uncertain future in the SBR. Decreases in timber harvest on public land have been implemented in the last five years. These decreases are likely to continue and may even culminate in a total ban on timber harvest on national forests. Such a ban may relegate the GWWA to a few scattered populations with poor viability, or may totally eliminate the GWWA from the SBR as has happened in the Great Smoky Mountain National Park.

The local extirpation of GWWA from the Great Smoky Mountain National Park suggests that natural disturbance regimes at current levels are unlikely to provide adequate habitat for this species in the SBR. Without allowing natural disturbance regimes, including intense wildfire, to return to historic levels, some form of human disturbance may be required to maintain GWWA in the SBR. Harvest intensities at expected levels may provide adequate habitat for some species, however the very small populations of GWWA in CNF suggest that additional efforts may be necessary to provide habitat for GWWA.

Limiting the harvest of timber also limits the management practices available to wildlife and forest managers. Focusing harvest on areas of high quality GWWA habitat may ensure the viability of this species in the region. To create high quality golden-winged warbler habitat our findings suggests that cuts should be clustered, located in oak-hickory or cove forest types, between 1000-1400 m in elevation. Stands between 3 and 13 yrs provide good nesting habitat for GWWA in the SBR. Harvests should be timed to provide stable levels of habitat. Every effort should be made to establish numerous

patches of herbaceous vegetation around the perimeter of the cut which should be mowed, burned, or regenerated with herbicides to maintain nesting sites. The mowing of roads and likely nesting sites should be delayed until after July 1st to avoid causing the nest failure of GWWA and numerous other species which nest in these habitats (e.g. black-throated blue warbler, chestnut sided warbler, rufous-sided towhee). This is critical since the colonial nature of GWWA may result in total loss of productivity if a colony is mowed during nesting. Driving need not be limited on roads with bare wheel ruts, however every effort should be taken to avoid driving on vegetated areas during the nesting season, including the grassy shoulders of rutted roads. Public access to these roads should be severely limited. Ideally the harvest of stands along an occupied road should be delayed until the completion of nesting since such operations often involve road grading and other improvements.

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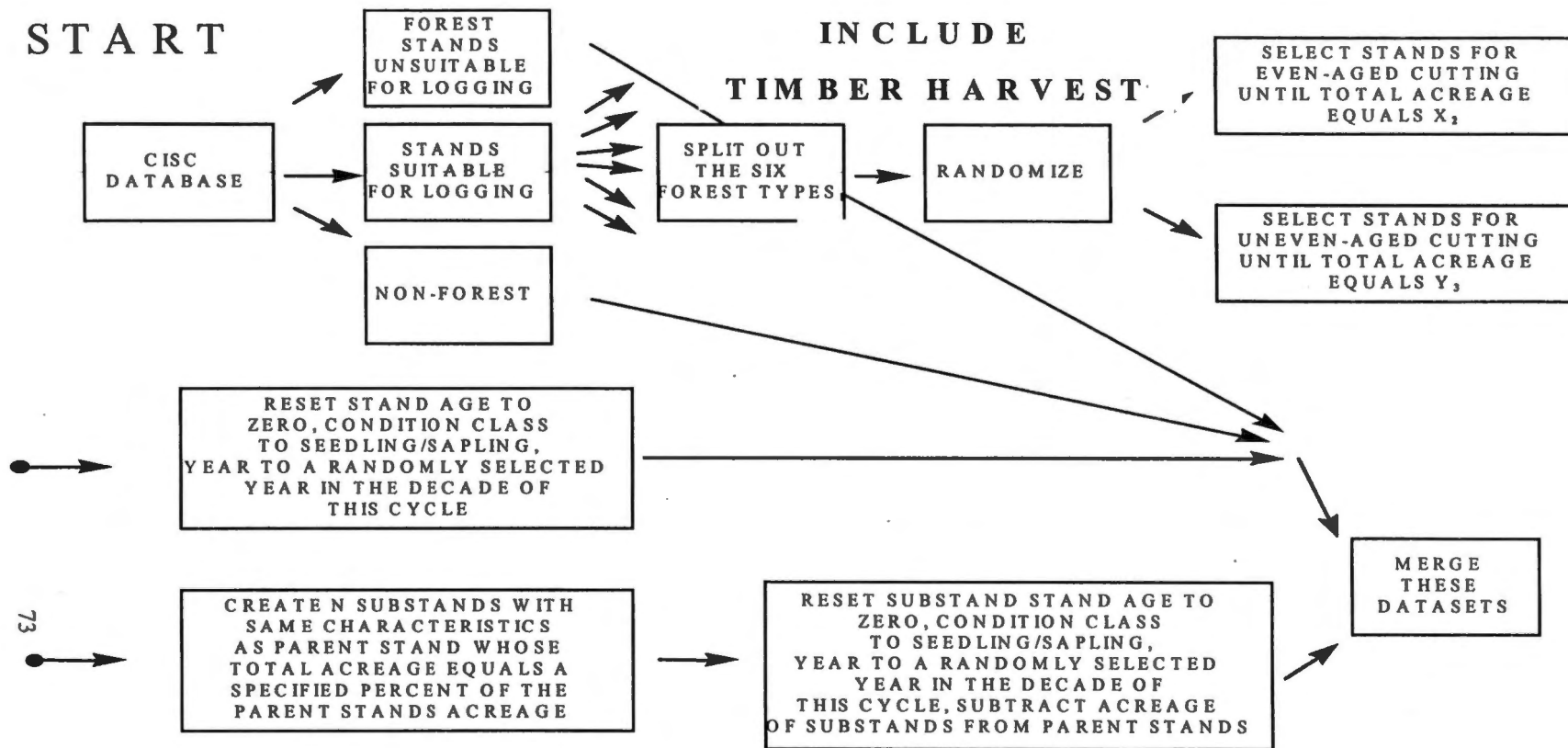
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APPENDICES

Appendix A.

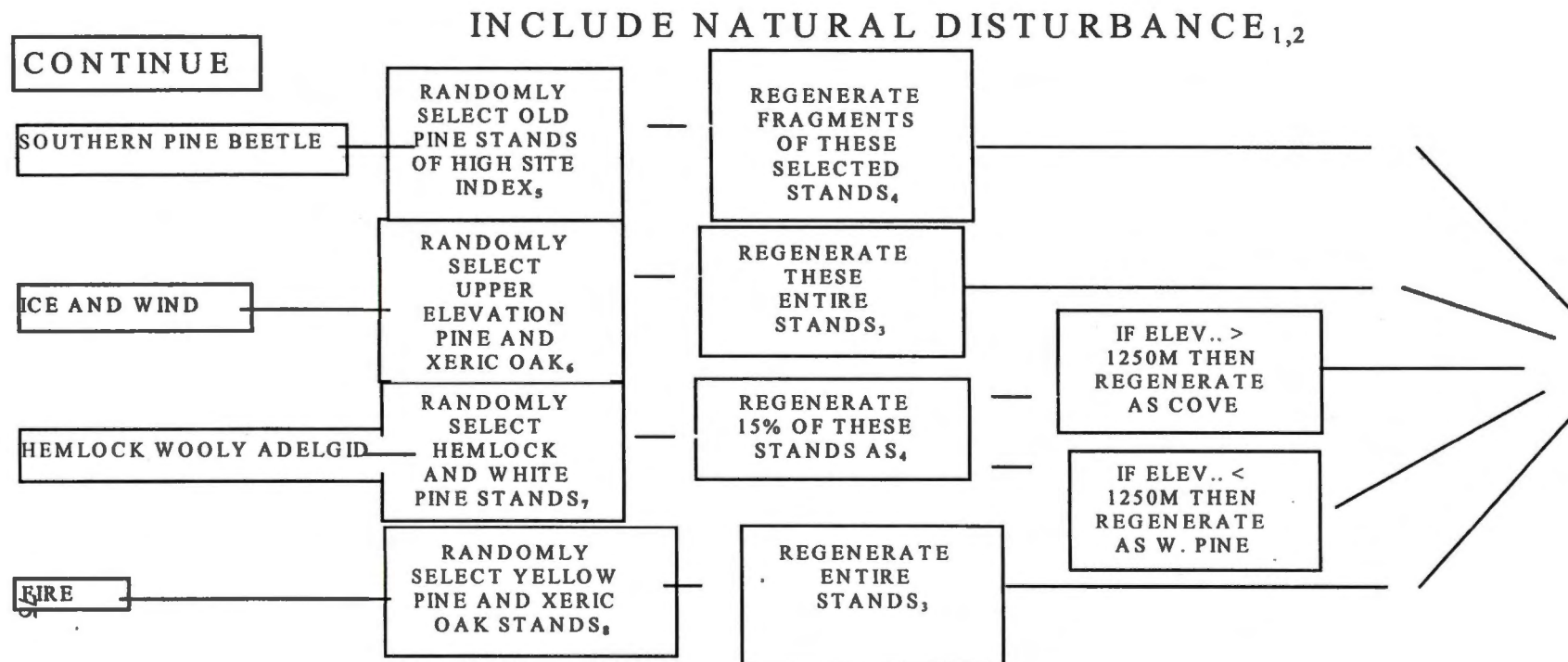
**A visual depiction of the forest model used to
forecast stand conditions and critical assumptions,
Cherokee National Forest, 1993-2053**



Appendix A. A flowchart depiction and underlying assumptions of the forest simulation model used to forecast stand conditions.

Appendix A. (continued)

1. We assumed that harvest regimes differ across the six broad forest types, but were the same within each forest type (Smith 1986).
The proportion of even-aged to uneven-aged harvests, area harvested, and several aspects of uneven-aged harvest varied by forest type.
2. We assumed that all even-aged management used clearcutting. Shelterwoods were infrequently used pre-1998, but were expected to play a larger role in the future. Current shelterwoods on the CNF had very low basal area, low residual canopy, and may have many similar bird habitat characteristics as clearcuts (Crawford et al. 1981, Annand and Thompson 1993).
3. We assumed that all uneven-aged harvests were group selection. We further assumed that harvested groups behaved as small clearcuts in terms of bird habitat. Group size and percent of stands harvested varied by forest type, ranging from 0.4 to 0.6 ha.

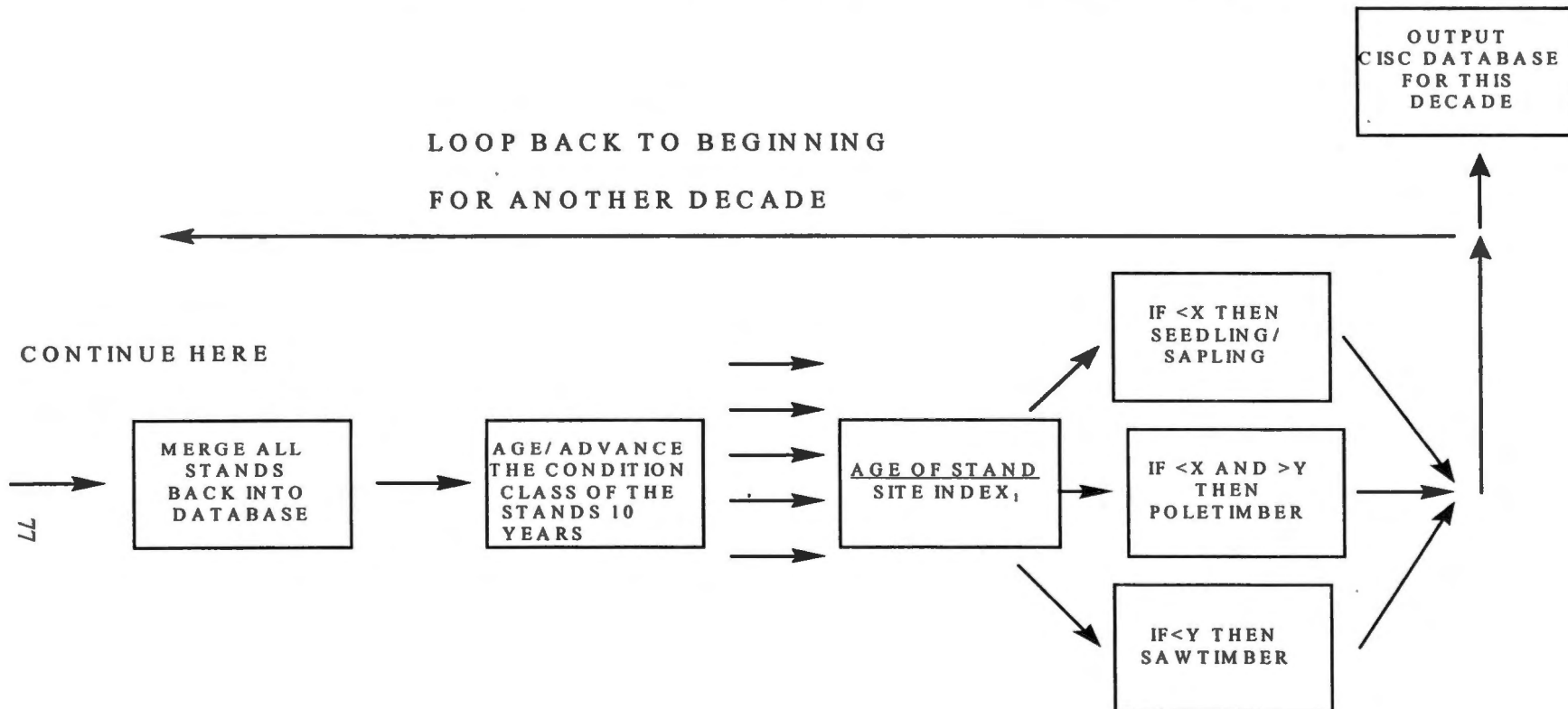


Appendix A. (continued)

Appendix A. (continued)

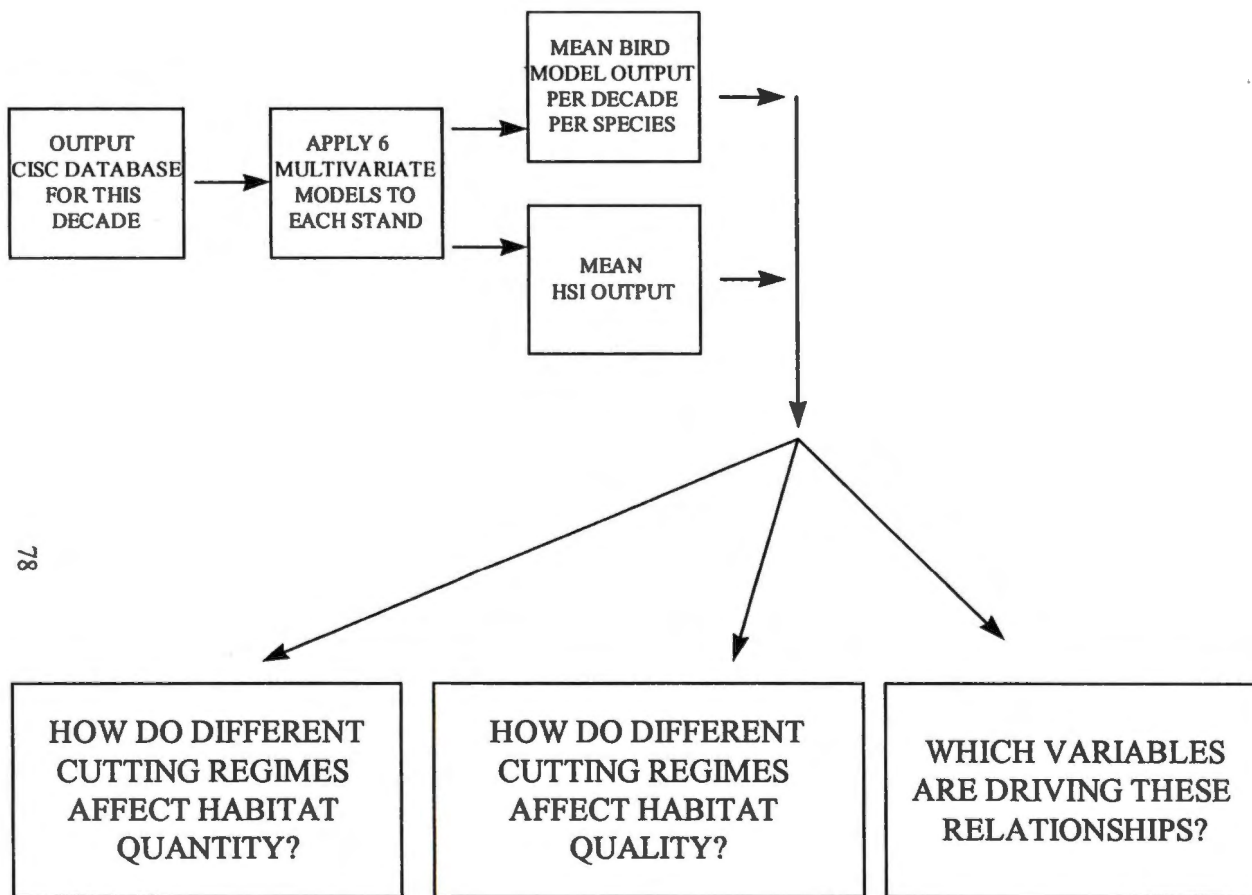
1. Assumed that natural disturbances operate at the stand level, and are random across the landscape within stands that meet criteria of past disturbances (Boerner et al. 1988).
2. Assumed that levels of natural disturbance will remain at similar to historic levels.
3. Assumed that regeneration caused by fire and hemlock woolly adelgid has the properties of clearcuts in terms of bird habitat.
4. Assumed that regeneration caused by southern pine beetle, ice, and wind has the properties of uneven-aged group selection in terms of bird habitat.
5. Southern pine beetle was predicted to infect 485 ha/10 years, killing 60% of trees in infected stands in 0.2 ha groups.
6. Ice and wind was predicted to damage 1012 ha/10 years, killing 35% of trees in damaged stands in 0.2 ha groups.
7. Hemlock woolly adelgid predicted to kill 121 ha/10 years kill all trees in each infected stand.
8. Fire expected to regenerate 152 ha/10 years, killing all trees in these stands.

ADVANCE THE AGE OF THE STANDS



Appendix A. (continued)

1. Advanced condition class based upon the ratio of stand age to site index. Cutoffs of this ratio for the three condition classes were specific to forest type and based upon ratios in the 1992 CISC database.



Appendix A. (continued)

Appendix B.

Factors influencing forest model outcome

Cherokee National Forest, Tennessee, 1993-2053

We identified three broad classes of natural disturbance which we felt included most natural disturbance that drove forest conditions over the last fifty years (fire, ice and wind, southern pine beetle), and one type of natural disturbance which will almost certainly influence forest conditions over the next fifty years (hemlock woolly adelgid). Two other effects also instrumental in forest modeling outcome are covered in this appendix: timber harvests and forest succession with time (Figure A1).

Fire

Fire has dominated the SBR over the past 10,000 years (Harmon 1982, SAMAB 1996, Delcourt and Delcourt 1996, Buckner and Turrell 1998) before colonization by Europeans. In contrast, over the last thirty years, fire has played an almost insignificant role in the SBR, and this state may continue into the immediate future. A successful campaign of fire suppression across the SBR has largely eliminated fire from the landscape. Over the last ten years, only 152 of 121,208 ha (0.12%) have experienced stand replacing fires in the southern half of CNF (R. Fryar, pers comm.). Nearly all of these fires have been limited to the yellow pine forest type. Today researchers are recognizing the importance of fire in the SBR; however public opinion is strongly against the use of fire in managing the SBR (SAMAB 1996). In spite of this, the USFS policy was changed in 1996 to allow non-threatening, naturally caused fires in designated wilderness areas to burn. However, such fires may be unlikely, since nearly any naturally caused fire will likely occur in dry conditions and will risk spreading to uncontrollable wildfire across the region. In addition, most designated wilderness areas contain little yellow pine or other fire-prone/fire-dependent communities. Such communities are

largely absent from much of their previous range because of decades of fire suppression. In our modeling scheme, we assumed stand-replacing fire frequency would continue to remain small. Our model causes approximately 150 ha of yellow pine or mixed pine hardwood forest types to experience stand-replacing fires every ten years. Our modeled burns were generally small and were scattered across the forest and across time as is the case today (SAMAB 1996). After these burns, stands were regenerated as yellow pine stands. Their ages were reset to the year of the burn.

Ice and Windstorm

Unlike wildfire, ice and wind damage are still frequent in CNF. Ice storms and wind events occur throughout the range of elevation and forest types on CNF, but the damage incurred drastically differs depending upon elevation, topography, forest type, and age (Carvell et al. 1957, Whitney 1982). The magnitude and frequency of damage in any one stand is considered unpredictable (Boerner et al. 1988). Ice and wind events were treated collectively in our model since damage caused by each depended upon similar characteristics of forest type, stand age, and elevation. In nature, ice damage is a function of the total surface area of a tree, the resiliency of its wood, the extent of its roots, and its position on a slope (Abell 1934, Carvell et al. 1957, Whitney 1982, Whitney and Johnson 1984, Boerner et al. 1988). In spite of numerous studies over the last century, there is little agreement about such basic facts as the frequency of damaging storms within the region. This is complicated by issues of scale and intensity; the definition of an ice storm varies widely between authors. For our purposes an ice storm was a weather event where glaze (a clear layer of ice formed by undercooled water

freezing on the surface of objects) formed upon vegetation such that significant and widespread damage was done to the canopy (Lemon 1961). Our definition of significant damage is a storm resulting in numerous canopy gaps at least 0.2 ha in size spread across ≥ 50 ha. In our study area, the mean number of days with glaze was six (Baldwin 1973); however, the frequency of severe ice damage in the SBR has been significantly less with estimates of one event every five to twenty years (Abell 1934, Bennett 1959). In the SBR, glaze is most common in higher elevations (Carvell et al. 1957); however, the incidence of significant damage is far more common in the mid-elevations due to tree species composition (Whitney and Johnson 1984, Boerner et al. 1988). In general, yellow pine, mixed, and oak/hickory are most susceptible to ice damage (Abell 1934, Carvell et al. 1957, Whitney 1982, Whitney and Johnson 1984, Boerner et al. 1988). Large trees are more susceptible than small flexible trees (Whitney 1982). Trees found on or near ridges are more susceptible to damage (Boerner et al. 1988). Whitney and Johnson (1984) found approximately 38 percent of trees in an ice damaged forest had died by the end of the second growing season. Our model anticipates a total of 1012 ha of mature yellow pine, mixed pine hardwood, and oak hickory stands to sustain ice or wind damage every ten years. Although our database contains no topographical component, it favors stands on ridges by selecting stands greater than 450 m elevation, the upper half in elevation of the yellow pine community. Damaged stands experienced regeneration in 38 percent of the stand, in groups of 0.2 ha (McKellar 1942, Whitney and Johnson 1984).

Hemlock Woolly Adelgid

The hemlock woolly adelgid is an Asian insect species first found in the eastern United States in 1924. The hemlock adelgid is believed to be spreading south along the Appalachians from the Shenandoah Valley (SAMAB 1996). This adelgid infects the eastern hemlock and Carolina hemlock, defoliating it and causing mortality within five years (McClure 1987). Hemlock populations will likely be decimated in the SBR. Though purely speculative, surveys of district silviculturists agreed that 75-100% of hemlock stands will be lost on CNF over the next fifty years. These stands will likely be replaced by cove hardwoods above 1250 m and by white pine below 1250 m (R. Fryar, pers. comm). Our model gradually replaced all hemlock stands between 1993 and 2053 at a rate of 121 ha/10 years. Hemlock stands were replaced with seedling/sapling cove hardwood above 1250 m and seedling/sapling white pine below 1250 m (R. Fryar, pers comm.). Hemlock woolly adelgid may substantially effect songbird abundance and diversity. A similar species, the balsam wooly adelgid, has already altered the structure and tree species composition of the southern Appalachians, resulting in significant changes in the bird community (Rabenold et al. 1997).

Southern Pine Beetle

Southern pine beetle (SPB) is an irregular source of disturbance in the SBR. Populations are cyclic within a locality but these cycles are not coordinated across the country (Price and Doggett 1978). Within a locality, populations cycle roughly every ten to twenty years. Population peaks, also known as epidemics, generally last two to three years (Payne 1981:7-27). During this time beetles attack and kill groups of yellow pines.

Shortleaf pine and pitch pine are the most common victims of SBP in the SBR, comprising 82 percent of all attacks though they comprise only 35 percent of yellow pines within the region. Ninety percent of SBP spots were less than 0.4 ha (Belanger 1981:165-176). The average SBP spot size in the southern CNF is approximately 0.2 ha (F. Locke, pers comm.). Stands with site indices above 80 are most at risk (Lorio and Sommers 1981:75-78). SPB attacks sawtimber-sized pines, especially those which are decadent. Our model included SPB spots on a twenty-year cycle. During such an epidemic, approximately 200 ha of shortleaf pine, pitch pine, and mixed yellow pine hardwoods were attacked with spot sizes of 0.2 ha killed, regenerating as seedling/sapling stands of the original forest type.

Timber Harvests

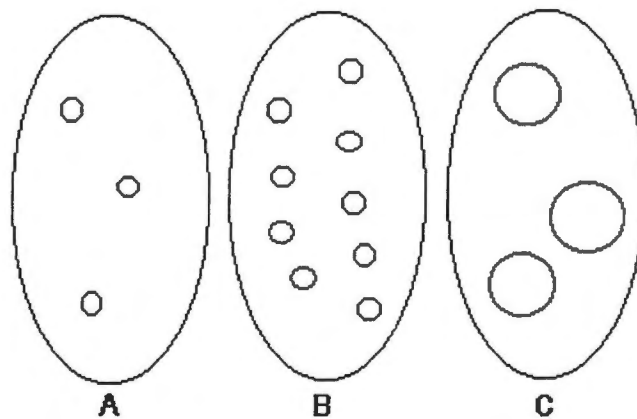
Timber harvest activities in CNF fall into two classes: even-aged and uneven-aged. Both even-aged and uneven-aged silviculture on CNF can be further subdivided into a variety of methods including clearcutting and shelterwood even-aged methods, and group or single tree selection uneven-aged methods. In addition, site preparations such as windrows, burning, and regeneration methods including natural regeneration, seed trees, and planting can also be used. Uneven-aged methods can additionally vary by average group size (area) and the percent of the total stand harvested (R. Fryar, pers comm.). Within CISC there are few variables available to model effects of various regeneration regimes. During the periods when point count data were collected to build bird habitat models, clearcutting was the dominant harvest method in CNF; shelterwood and uneven-aged methods were virtually absent from the landscape. As a result, our bird habitat

models did not include a harvest type variable. Stand age and condition class adequately encompassed the impacts of timber harvests upon birds at the time. Thus, to simulate harvests of any type upon bird habitat, we were limited to the five CISC variables used in the bird models; any timber harvest modeled must exist as some form of clearcut. Consequently, our model of harvest activities was far more simplistic than reality. In any integration of forest models and wildlife habitat models one faces a compromise between those variables best suited for monitoring forests on a landscape scale and those variables best suited to predicting wildlife habitat (Smith 1986).

In spite of these difficulties, our timber harvest models included simulations of both even and uneven-aged methods. Our model of even-aged harvests was limited to clearcuts. We could not model shelterwoods since such a stand had no single value for stand age or condition class. This difficulty may not be as serious as it appears. Most USFS shelterwoods differ dramatically from a classic shelterwood. In USFS shelterwoods, residual basal area is usually very low; the canopy is removed and only scattered trees remain. As a result regeneration is very dense. Such shelterwoods appear to be very similar to clearcuts from the standpoint of bird habitat. Lacking a better alternative, shelterwood cuts were lumped together with clearcuts in our model. Care should be taken when considering the relative impacts of true shelterwoods upon bird habitat. Acreage harvested using even-aged techniques was specific to each forest type and based upon the expert opinions of district silviculturists of CNF.

Although complicated from a programming standpoint, the inclusion of uneven-aged harvest was possible. All simulated uneven-aged harvests were in the form of group selection. Single tree selection could not be included in our modeling program for

reasons that will become apparent. To adapt group selection to the various regeneration needs of different forest types, group size and percent of a stand harvested were variable. To distinguish harvested groups from the remaining older stands, each cut group became a new stand after harvest, with characteristics identical to the 'parent' stand except that the area of each new stand was set to the average group size of harvests in that forest type; the group's year was reset to the year of harvest; and the condition class was set to seedling/sapling. Essentially each harvested group became a miniature clearcut. When a parent stand was harvested using group selection, successive groups were removed from the acreage of the parent stand until the sum of these groups was equal to the percentage of a parent stand typically harvested by group selection. This percentage varied between forest types and was specific in our model to each forest type.



Above: An illustration demonstrating group size and percent of parent stand harvested. Stands A and B are harvested using group selection with equal group size. Stand B has a larger percent of its parent stand harvested than A. Stands C and B have equal percents of their parent stands harvested; however, stand C has a larger group size than stand B.

Forest Succession

Forest succession played a major role in our model results. Forest succession influenced the stand age and condition class variables. Stand age was calculated by subtracting the date the stand was last harvested from the current year being modeled. Condition class was also updated through time. Condition class was calculated for each forest type separately since growth rates vary by forest type and site index. Condition class was calculated by obtaining the age fraction, the ratio of the age of the stand in the currently modeled year to the site index. Confidence intervals were calculated for each combination of forest type and condition class in the original 1993 CISC database. These age fraction confidence intervals then defined cutoffs for updating condition class in each decade.

Model Limitations

Cherokee National Forest was not in a steady state in terms of the stand age distribution during our simulations and will not be for many years. Changes in age structure, species composition, and disturbance regimes will undoubtedly affect the forest in many different and perhaps unforeseeable ways. Perhaps the greatest difficulty we encountered from this instability were increased influences of stand age in some of our bird habitat models. Every passing year caused changes in CNF which make our bird habitat models less accurate. Eventually this led to serious biases in bird models where stand age was a continuous variable. As with any forecasting exercise the accuracy of our predictions decrease as the further into the future we looked.

A second omission from our forest model was the absence of any gypsy moth component. If the gypsy moth reaches Cherokee National Forest, it will likely dominate changes in forest structure and composition for decades. However this radically different future is far from certain because gypsy moth may not reach Cherokee National Forest. There may be some natural limit to gypsy moth range, or human intervention may stop this destructive exotic. In the face of this uncertainty we felt our best option was to simply plunge forward in spite of the gypsy moth uncertainty.

Appendix B. - Input variables and areas affected in the forest stand model, southern districts Cherokee National Forest.

Variable	Hectares affected/ 10 years at expected harvest levels
Hectares of clearcutting in y. pine	404
Hectares of group selection harvest in y. pine	303
percent of stand cut in group selection in y. pine	30%
size of groups used in group selection in y. pine	0.4
Hectares of clearcutting in white pine/hemlock	579
Hectares of group selection harvest in white pine hemlock	65
percent of stand cut in group selection in pine white pine/hemlock	30%
size of groups used in group selection in white pine/hemlock	0.4
hectares of clearcutting in cove hardwoods	197
hectares of group selection harvest in cove hardwoods	197
percent of stand cut in group selection in cove hardwoods	30%
size of groups used in group selection in cove hardwoods	0.4
hectares of clearcutting in mixed hardwood/pine	384
hectares of group selection harvest in mixed hardwood/pine	20
percent of stand cut in group selection in mixed hardwood/pine	30%
size of groups used in group selection in mixed hardwood/pine	0.4
hectares of clearcutting in northern hardwoods	0
hectares of group selection harvest in northern hardwoods	0
percent of stand cut in group selection in northern hardwoods	0%
size of groups used in group selection in northern hardwoods	0

Appendix B. (continued)

Variable	hectares affected/ 10 years
hectares of clearcutting in oak/hickory	546
hectares of group selection harvest in oak/hickory	546
percent of stand cut in group selection in oak/hickory	30%
size of groups used in group selection in oak/hickory	.61
hectares of stand regenerating burns	152
hectares of stands infected with southern pine beetle	970/20 years
% of infected stands killed	60%/20 years
average size of pine beetle spot	0.2/20 years
hectares of hemlock stands killed by wooly adelgid	121
hectares disturbed by ice/windthrow	1012
% of disturbed stands killed by ice/windthrow	35%
patch size of ice disturbance	0.2

Appendix C.

Common and scientific names of plant species

Appendix C. Common and scientific names of plant and animal species, southern districts, Cherokee National Forest, Tennessee, and Nantahala National Forest, North Carolina.

Common Name	Scientific Name	Common Name	Scientific Name
Alder	<i>Alnus spp.</i>	mountain laurel	<i>Kalmia latifolia</i>
alternate leaf dogwood	<i>Cornus alternifolia</i>	mountain maple	<i>Acer spicatum</i>
aspen	<i>Populus spp.</i>	rhododendron	<i>Rhododendron maximum</i>
basswood	<i>Tilia americana</i>	multiflora rose	<i>Rosa multiflora</i>
black birch	<i>Betula lenta</i>	northern red oak	<i>Quercus rubra</i>
black cherry	<i>Prunus serotina</i>	pignut hickory	<i>Carya glabra</i>
black gum	<i>Nyssa sylvatica</i>	pitch pine	<i>Pinus rigida</i>
black haw	<i>Viburnum spp.</i>	post oak	<i>Quercus stellata</i>
black locust	<i>Robinia pseudo-acacia</i>	red maple	<i>Acer rubrum</i>
black oak	<i>Quercus velutina</i>	redbud	<i>Cercis canadensis</i>
blackberry	<i>Rubus spp.</i>	sassafras	<i>Sassafras albidum</i>
blackjack oak	<i>Quercus marilandica</i>	scarlet oak	<i>Quercus coccineata</i>
blueberry	<i>Vaccinium spp.</i>	scarlet oak	<i>Quercus coccineata</i>
Carolina buckthorn	<i>Rhamnus caroliniana</i>	shortleaf pine	<i>Pinus echinata</i>
Carolina hemlock	<i>Tsuga caroliniana</i>	silverbell	<i>Halesia carolina</i>
chestnut oak	<i>Quercus prinus</i>	sourwood	<i>Oxydendrum arboreum</i>
crossvine	<i>Bignonia capreolata</i>	Southern red oak	<i>Quercus falcata</i>
cucumber tree	<i>Magnolia acuminata</i>	spicebush	<i>Lindera bezoin</i>
dog hobble	<i>Leucothoe fontanesiana</i>	striped maple	<i>Acer pensylvanica</i>
downy serviceberry	<i>Amelanchier arborea</i>	sugar maple	<i>Acer saccharum</i>
eastern hemlock	<i>Tsuga canadensis</i>	table mountain pine	<i>Pinus pungens</i>
eastern white pine	<i>Pinus strobus</i>	tamarack	<i>Larix spp.</i>
farkleberry	<i>Vaccinium arboreum</i>	tulip poplar	<i>Liriodendron tulipifera</i>
fire cherry	<i>Prunus pensylvanica</i>	vaccinium	<i>Vaccinium spp.</i>
Fraser fir	<i>Abies fraseri</i>	Virginia creeper	<i>Parthenocissus quinquefolia</i>
goldenrod	<i>Solidago spp.</i>	Virginia pine	<i>Pinus virginiana</i>
grape	<i>Vitis spp.</i>	white oak	<i>Quercus alba</i>
greenbriar	<i>Smilax spp.</i>	wild hydrangea	<i>Hydrangea arborescens</i>
hobblebush	<i>Viburnum alnifolium</i>	yellow birch	<i>Betula lutea</i>
mockernut	<i>Carya tomentosa</i>		
hickory			

Appendix D

Common and scientific names of animal species and other miscellaneous species

Appendix D. Common and scientific names of animal and miscellaneous species.

Common Name	Scientific Name	Common Name	Scientific Name
Bird Species		Mammal Species	
Acadian flycatcher	<i>Epidonax virescens</i>	grizzly bear	<i>Ursus arctos</i>
black-backed woodpecker	<i>Picoides articus</i>	northern flying squirrel	<i>(Glaucomys sabrinus coloratus)</i>
blue-headed vireo	<i>Vireo solitarius</i>	Insect Species	
blue-winged warbler	<i>Vermivora pinus</i>	balsam woolly adelgid	<i>Adelges piceae</i>
brown-headed cowbird	<i>Molothrus ater</i>	hemlock wooly adelgid	<i>Adelges tsugae</i>
chestnut-sided warbler	<i>Dendroica pensylvanica</i>	southern pine beetle	<i>Dendroctonus frontalis</i>
golden-winged warbler	<i>Vermivora chrysoptera</i>	Fungi species	
three-toed woodpecker	<i>Picoides tridactylus</i>	chestnut blight	<i>Cryphonectria parasitica</i>
tufted titmouse	<i>Parus bicolor</i>		
yellow-billed cuckoo	<i>Coccyzus americanus</i>		
yellow-throated warbler	<i>Dendroica dominica</i>		

Appendix E
Tables and Figures

Table 1. Hectares of forest (suitable and unsuitable for timber management) by forest type and age class, southern districts, Cherokee National Forest, Tennessee, 1992.

Forest Type	Suitable Hectares			Unsuitable Hectares				Totals
	0-20	21-60	>60	0-20	21-60	>60	percent	
	years	years	years	years	years	years	unsuitable	
Cove Hardwood	741	629	7522	80	381	5890	41.2	15243
Eastern Hemlock / White Pine	898	1862	1806	14	164	1362	25.2	6106
Mixed Hardwood/ Pine	1206	2641	14850	18	512	11812	40.0	31039
Northern Hardwood	94	53	211	0	150	1203	79.1	1711
Oak / Hickory	754	1448	12145	14	515	11473	46.0	26349
Yellow Pine	4021	7281	10042	164	975	6604	27.0	29087
TOTALS	7714	13914	46576	290	2697	38344	avg=43.0	109535

Table 2. Harvest intensity by forest type from 1942-1992 in the southern districts,
Cherokee National Forest, Tennessee.

Forest Type	Total Hectares	Sum Hectares cut 1942-1992	% of Total Harvested 1942-1992
Cove Hardwoods	15244	1456	10
E Hem/W Pine	6107	2758	45
Mixed Hwood/Pine	31040	3668	12
Northern Hardwood	1712	297	17
Oak/Hickory	26350	2027	8
Yellow Pine	29089	11654	40

Table 3. Habitat variables and descriptions used to construct bird habitat models, southern districts, Cherokee National Forest.

CISC	Model	Variable	Description	Range
Variable	Variable	Type		
Age_year	stnd. age	discrete	current year - year of last harvest or catastrophic disturbance	0-∞ (0-172 in 1999)
Fortype	y. pine	binary	yellow pine forest type	0 (no) - 1 (yes)
	wp/hem	binary	white pine or hemlock forest type	0 (no) - 1 (yes)
	cove	binary	cove hardwood forest type	0 (no) - 1 (yes)
	n.hard	binary	northern hardwood forest type	0 (no) - 1 (yes)
	mixed	binary	mixed pine hardwood forest type	0 (no) - 1 (yes)
	oak/hic	binary	oak/ hickory forest types	0 (no) - 1 (yes)
Standcon	seed	binary	seedling/sapling condition class	0 (no) - 1 (yes)
	pole	binary	poletimber condition class	0 (no) - 1 (yes)
	saw	binary	sawtimber condition class	0 (no) - 1 (yes)

Table 3. (continued)

CISC	Model	Variable	Description	Range
Variable	Variable	Type		
S_index	site index	discrete	site potential, dom. tree ht in 50 years	40-130
	site index1	binary	site index <70	0 (no) - 1 (yes)
	site index2	binary	70 < site index <80	0 (no) - 1 (yes)
	site index3	binary	80 < site index < 110	0 (no) - 1 (yes)
Elevation	elev	continuous	elevation in meters	231-1530
	elev1	binary	elev >475	0 (no) - 1 (yes)
	elev2	binary	475 < elev < 872	0 (no) - 1 (yes)
	elev3	binary	618 < elev < 872	0 (no) - 1 (yes)
	elev4	binary	elev <872	0 (no) - 1 (yes)

Table 4. Mean breeding densities, standard error, and breeding density ranges for the six modeled species based upon breeding bird census data (Hamel 1992).

Species	Breeding Density (pairs/40 ha)			Density Range	
	Mean	S.E.	N	Low	High
Acadian flycatcher	14.5	1.45	102	1	43
blue-headed vireo	8.6	2.3	24	3	28
chestnut-sided warbler	11.9	3.4	13	6	36
tufted titmouse	12.1	0.7	143	1	23
10 yellow-billed cuckoo	6.1	0.6	101	0.4	8.5
yellow-throated warbler	5.1	0.91	41	1.7	25

Table 5. Bird habitat models and associated statistics for six CISC based bird habitat models, Cherokee National Forest, Tennessee, 1998.

AOU spp. Code	Variables Selected	Parameter Estimate	Pr. $>\chi^2$	Max- rescaled R-square	Hosmer – Lemeshow Statistic P – value	Concordance
ACFL	Intercept	-1.5637	0.0796	0.3688	0.4509	85.0
	elev (m)	-0.0014	0.0002			
	Hem/wp	0.3419	0.6260			
	oak/hic	-1.1670	0.2371			
	Cove	0.2327	0.7444			
	n. hard	1.0615	0.3419			
	Mixed	-1.1889	0.1310			
	stnd. Age	0.0487	0.0001			
BHVI	Intercept	-2.1232	0.2721	0.5559	0.8779	88.2
	elev (m)	0.0019	0.0001			
	hem/wp	0.6052	0.5253			
	oak/hic	0.3936	0.6252			
	Cove	1.0873	0.0821			
	n. hard	0.8187	0.3964			
	Mixed	0.3505	0.6448			
	stand age1	-2.5407	0.0001			
	stand age2	-0.4557	0.4398			
	stand age3	-1.1365	0.0424			
CSWA	site index	-0.0412	0.1233	0.6484	0.6616	95.6
	Intercept	-9.1513	0.0098			
	elev (m)	0.0022	0.0002			
	Seed	3.5182	0.0001			
	hem/wp	0.7384	0.5249			
	oak/hic	0.6804	0.6117			
	Cove	0.5781	0.7677			
	n. hard	1.1052	0.4114			
	site index	-0.0188	0.6961			

Table 5. (continued)

AOU spp. code	Variables Selected	Parameter Estimate	Pr. >χ^2	Max- rescaled R-square	Hosmer – Lemeshow Statistic P – value	Concordance
TUTI	intercept	-5.2213	0.0001	0.2508	0.9997	76.7
	elev1	2.8392	0.0033			
	elev2	2.1794	0.0247			
	elev3	1.6201	0.0714			
	hem/wp	1.8272	0.0157			
	oak/hic	1.0733	0.1828			
	cove	3.1411	0.0014			
	n. hard	2.3897	0.0428			
	seed	-0.7605	0.0933			
	pole	-0.9016	0.0461			
	site	2.9730	0.0031			
	index1					
	site	0.6899	0.3430			
	index2					
YBCU	intercept	1.2637	0.0538	0.2492	0.8464	78.3
	elev (m)	-0.00138	0.0001			
	oak/hic	1.0905	0.0670			
	mixed	0.7848	0.0735			
	stand age	-0.00883	0.2055			
YTW A	intercept	1.9322	0.2774	0.2051	0.9541	78.0
	elev (m)	-0.0007	0.0211			
	hem/wp	1.1849	0.1091			
	oak/hic	-0.5287	0.5472			
	Cove	0.5857	0.5546			
	Mixed	0.4610	0.4244			
	stand age	0.0266	0.0023			
	site index	-0.0578	0.0455			

Table 6. Results of a sensitivity analysis of forest simulation variables on bird habitat units for the six modeled species. P-values ≤ 0.05 indicate that a 30% variation caused significant differences in available habitat units for that species.

Forest type	Distur.	Variable	acfl F	p	bhvi F	p	cswa F	p	tuti F	p	ybcu F	p	ytwa F	p
Yellow	even	annual harv.	234.83	0.0001	291.55	0.0001	2.91	0.0679	41.69	0.0001	97.18	0.0001	201.07	0.0001
Pine	uneven	annual harv.	17.58	0.0001	18.3	0.0001	2.72	0.0803	3.37	0.0464	9.32	0.0006	15.16	0.0001
Hemlock/	even	annual harv.	428.53	0.0001	197.97	0.0001	1.94	0.1587	150.05	0.0001	190.77	0.0001	315.3	0.0001
W. pine	uneven	annual harv.	1.21	0.3105	1.38	0.2659	1.88	0.1684	3.38	0.0458	0.35	0.707	0.98	0.3863
Mixed	even	annual harv.	360.13	0.0001	224.17	0.0001	7.18	0.0025	63.06	0.0001	393.55	0.0001	434.2	0.0001
Ypine/hw	uneven	annual harv.	0.81	0.4527	0.02	0.977	2.43	0.1035	2.31	0.1143	0.23	0.7951	0.59	0.5584
Cove	even	annual harv.	128.69	0.0001	39.11	0.0001	33.59	0.0001	45.99	0.0001	23.83	0.0001	53.48	0.0001
Hardwd	uneven	annual harv.	24.76	0.0001	10.21	0.0003	0.24	0.7897	6.76	0.0034	9	0.0007	13.53	0.0001

Table 6. (continued)

Forest type	Distur.	Variable	acfl F	p	bhvi F	p	cswa F	p	tuti F	p	ybcu F	p	ytwa F	p
Northern	even	annual harv.	0.19	0.8259	0.18	0.8384	0.12	0.8838	0.24	0.7859	0.21	0.8085	0.19	0.8243
Hardwd	uneven	annual harv.	0.22	0.8041	0.22	0.8041	0.22	0.8041	0.22	0.8041	0.22	0.8041	0.22	0.8041
Oak/hick.	even	annual harv.	385.41	0.0001	628.45	0.0001	43.72	0.0001	82.23	0.0001	298.47	0.0001	203.61	0.0001
	uneven	annual harv.	36.14	0.0001	92.79	0.0001	15.25	0.0001	8.23	0.0012	24.72	0.0001	16.22	0.0001
Y. pine	south.	area affect.	0.28	0.7578	0.27	0.7621	1.03	0.3698	0.16	0.8551	0.09	0.9105	0.38	0.6848
And	pine	spot size	0.99	0.3858	0.57	0.572	0.23	0.7958	0.37	0.6961	1.02	0.3751	0.9	0.4168
Mixed	beetle	% kill	0.06	0.9444	0.17	0.8417	0.78	0.6095	0.27	0.7657	0	0.9967	0.09	0.9182
Yp., m, O	fire	area burnt	1.53	0.2341	1.21	0.3139	0.50	0.6095	1.1	0.3475	1.36	0.2731	1.33	0.2794
Oak/hic,	ice and	area affect.	50.23	0.0001	81.13	0.0001	42.84	0.0001	38.56	0.0001	43.02	0.0001	49.23	0.0001
Y.p., mix	wind	patch size	47.2	0.0001	86.66	0.0001	1.32	0.2828	57.72	0.0001	35.69	0.0001	48.14	0.0001
		% kill	0.41	0.6663	0.32	0.7267	13.71	0.0001	0.41	0.6661	0.31	0.7329	0.07	0.9312
Hemlock	H. Adel	area killed	3.35	0.0486	1.65	0.2095	0.98	0.3860	0.98	0.3857	3.67	0.0377	2.29	0.1187

Table 7. A summary of GWWA habitat sampling methods used in Cherokee National Forest, Tennessee and Nantahala National Forest, North Carolina, 1997-1998.

Variable Category	Sampling Method	Variable	How Measured
Trees	USFS records	Forest Type	Numeric code
Trees	USFS records	Stand Age	Numeric code
Trees	Prism	Basal Area (m ² /ha)	2.5 factor prism
Trees	Prism	Species	Scientific Name
Trees	Prism	DBH (cm)	DBH tape
Trees	Prism	Tree Height (m)	Clinometer and 50m tape
Trees	Point Sampling	Canopy Cover (%)	Spherical densiometer, 4 cardinal directions
Saplings	0.00314 ha plots	Species	Scientific Name
Saplings	0.00314 ha plots	Sapling Stem Count	By species
Saplings	0.00314 ha plots	Sapling Height (m)	Average for each species
Seedlings	0.00314 ha plots	Dominant Species	Scientific Name
Seedlings	0.00314 ha plots	Seedling Count (#/ha)	Sum over all spp.
Shrubs	0.00314 ha plots	Species	Scientific Name
Shrubs	0.00314 ha plots	Shrub Count (#/ha)	By Species
Vines	0.00314 ha plots	Species	Scientific Name
Vines	0.00314 ha plots	Vine Stem count (#/ha)	By Species
Vines	0.00314 ha plots	Vine Avg. Height (m)	By Species
Herbaceous vegetation	0.00314 ha plots	Average Height (m)	Across all species
Herbaceous vegetation	0.00314 ha plots	Dominant Species	Species names of 3 w/greatest cover
Herbaceous vegetation	0.00314 ha plots	Percent Cover (%)	For 3 most common, ocular
Herbaceous vegetation	0.00314 ha plots	Density (1-5)	Classes 1-5, 5 highest
Herbaceous vegetation	0.00314 ha plots	Percent grass (%)	Percent of forb volume in grass
Physical Characteristic	Point Sampling	Distance to nearest road (m)	Laser range finder
Physical Characteristic	Point Sampling	Road Type (1-5)	Classes 1-5, 5 most used

Table 7. (continued)

Variable Category	Sampling Method	Variable	How Measured
Physical Characteristic	Point Sampling	Slope (°)	Clinometer to nearest degree
Physical Characteristic	Point Sampling	Aspect (°)	Compass to nearest degree
Physical Characteristic	Point Sampling	Elevation (m)	Off USGS maps

Table 8. Locations, habitats used, and approximate numbers of GWWA territories in the Nantahala and Cherokee National Forests, North Carolina and Tennessee, 1998.

Region	Habitat(s) Used	Number of Territories
Wachacha Bald Area, NC	complex of at least seven cuts	31
Bert's Creek/Frank's Creek, NC	complex of at least seven cuts	11
Big Choga Area, NC	complex of at least eight cuts	8
Stecoah Gap Area, NC	three cuts, plus large roadside area	11
scattered individuals, NC	roadside, wildlife openings, powerline rights of way, clearcuts, wetlands	15
scattered individuals, TN	Clearcuts, group selection	5
Total		81

Table 9. Habitat characteristics of forest stands occupied (N = 31) and unoccupied (N = 26) by golden-winged warblers, Cherokee National Forest, Tennessee, and Nantahala National Forest, North Carolina.

Variable	Occupied			Unoccupied			Logistic Regression Parameter estimate	Wald-chi Square P
	Median	Min.	Max.	Median	Min.	Max.		
Intercept							-1.4774	0.7776
Basal area (m ² /ha)	10	0	35	40	0	48	-0.0482	**0.0310
Roadtype (1-5)	3	1	5	3	1	5	1.0571	**0.0332
Num. Sap. Stems/ha	4445	0	12446	5271	0	32258	0.0301	*0.0557
Average ht. trees (m)	10.8	0	18.9	10.20	0	20	0.2101	*0.0655
Percent grass ¹	2	0	20	0	0	60	0.2083	0.1091
Blackberry ht. ¹ (m)	1.0	0	2	0.6	0	2	1.5050	0.1796
Elevation (m)	1066.8	731.5	1341.1	1097.28	792.5	1280.2	-0.00135	0.2864
Slope (degrees)	25	13	37	22.5	0	40	-0.0485	0.3780
Canopy cover var. ²	19.7	1.63	37.5	9.52	3.4	46.7	0.0370	0.5052
Herb. Diversity ¹	20.2	0	57.7	25.35	0	53.7	-0.0150	0.6281
Aspect (degrees)	190	27	352	220	5	325	-0.00205	0.6845
Concordance = 93.5%								
Max-rescaled R-squared = 0.7037								
Goodness-of-Fit Statistic = 0.9299 with 7 DF (P = 0.9959)								

¹ Estimated using a 5 m fixed radius plot.

² Std. error of four measurements taken in the cardinal directions 10 m from plot center

Table 10. Habitat characteristics of nest sites (N = 22) and surrounding points (N = 88) of golden-winged warblers, Cherokee National Forest, Tennessee, and Nantahala National Forest, North Carolina.

Variable	Nest			Surrounding pts.			Logistic Regression Parameter estimate	Wald-chi Square P
	Median	Min.	Max.	Median	Min.	Max.		
Intercept							-6.9576	0.0360
Herb. Density (1-5)	4	1	5	4	3	5	1.1524	**0.0097
Percent grass ¹	15	0	86	5	1	85	0.0194	0.1276
Canopy cover var. ²	25.6	0	49.0	21.4	2.4	50.6	0.0341	0.1532
Basal area (m ² /ha)	5	0	20	7.5	0	25	-0.0495	0.2380
Blackberry ht. ¹ (m)	1.3	0	3	1.2	0	2.5	0.3182	0.4886
Herb. Diversity ¹	28.3	0	56.9	20.8	4.6	53.7	0.0071	0.7017
Average ht. trees (m)	8.3	0	28	10.6	0	20	0.0276	0.5972
Num. Sap. Stems/ha	889	0	22860	2223	0	25400	0.0016	0.8601

Concordance = 79.8%

Max-rescaled R-square = 0.2930

Goodness-of-Fit Statistic = 9.9232 With 8 DF (P = 0.2705)

¹Measured using a 5 m fixed radius plot.

²Std. error of four measurements taken in the cardinal directions 10 m from plot center

Table 11. Habitat characteristics of nest sites (N = 22) and territories (N = 31) of golden-winged warblers, Cherokee National Forest, Tennessee, and Nantahala National Forest, North Carolina.

Variable	Nests			Territories			Logistic Regression Parameter estimate	Wald-chi Square P
	Median	Min.	Max.	Median	Min.	Max.		
Intercept							0.2670	0.7870
Num. Sap. stems/ha	889	0	25400	4445	0	32258	-0.0182	**0.0228
Avg. canopy cov. ¹	38.75	5	73.8	63.75	5	97.5	-0.0333	**0.0327
Canopy cover var. ²	25.6	2.4	50.6	19.7	3.4	46.7	0.0586	0.0687
Blackberry stems/ha	150	0	2258	44	0	600	0.0012	0.3552
Concordance = 82.6%								
Max-rescaled R-square = 0.4287								
Goodness-of-Fit Statistic = 5.8925 With 9 DF (P = 0.7506)								

¹Measured using a 5 m fixed radius plot.

²Std. error of four measurements taken in the cardinal directions 10 m from plot center

Table 12. Habitat characteristics of forest stands unoccupied (N = 26) and nest sites (N = 22) by golden-winged warblers, Cherokee National Forest, Tennessee, and Nantahala National Forest, North Carolina.

Variable	Nest site			Unoccupied			Logistic Regression Parameter estimate	Wald-chi Square P
	Median	Min.	Max.	Median	Min.	Max.		
Intercept							1.9998	
Blackberry stems/ha	18986	0	286766	635	0	78994	0.0009	0.4907
Num. Sap. Stems/ha	889	0	25400	5271	0	12446	-0.0147	0.2621
Road type (1-5)	3	1	5	3	1	5	0.6067	0.2164
Average ht. trees (m)	8.3	0	20	10.2	0	18.9	-0.0960	0.2460
Blackberry ht. ¹ (m)	1.3	0	2.5	0.6	0	2	1.0927	0.1926
Elevation (m)	1021.1	731.5	1280.2	1097.3	731.5	1341.1	-0.0029	0.3776
Slope (degrees)	27	0	40	22.5	13	37	-0.0564	0.2426
Canopy cover var. ²	25.6	2.4	50.6	9.5	1.6	37.5	0.1074	0.0075
Herb. Diversity ¹	28.3	4.6	53.7	25.3	0	57.7	-0.0120	0.6406
Aspect (degrees)	126	0	348	220	27	352	-0.0073	0.1543
Concordance = 91.6								
Max-rescaled R-squared = 0.6523								
Goodness-of-Fit Statistic = 8.8557 with 8 DF (P = 0.3546)								

¹Measured using a 5 m fixed radius plot.

²Std. error of four measurements taken in the cardinal directions 10 m from plot center

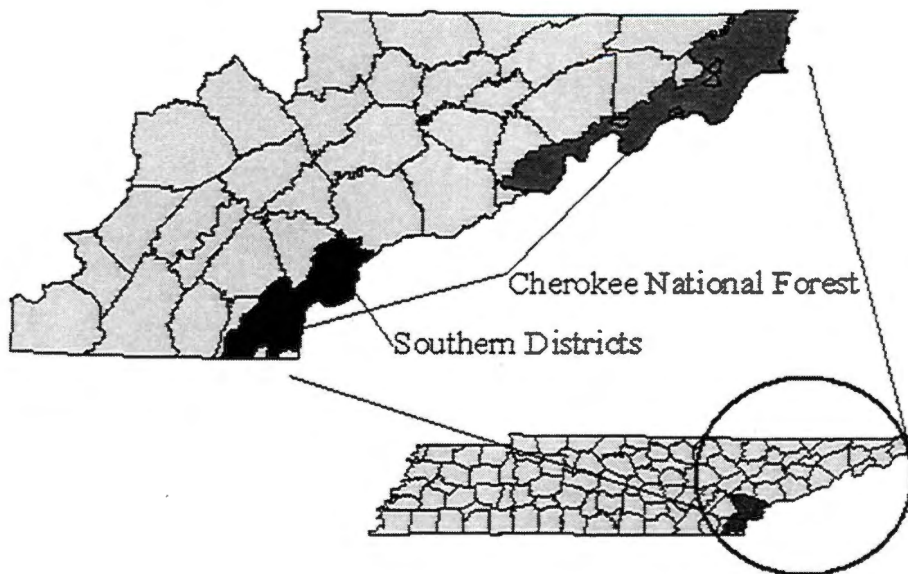


Figure 1. Location of Cherokee National Forest in eastern Tennessee. Our study area was the southern portion of the forest shown in black.

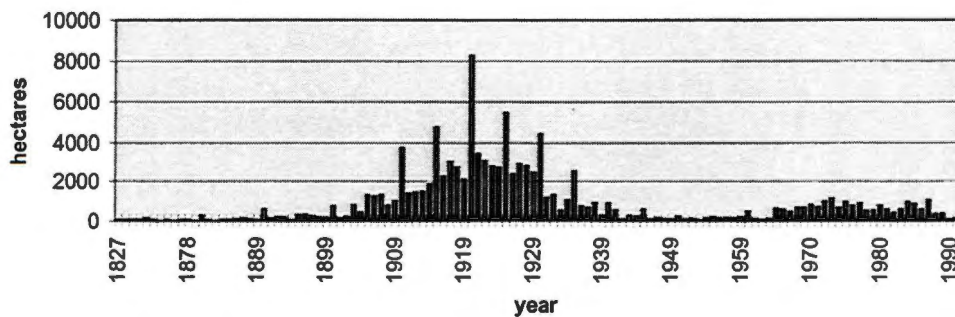


Figure 2. A frequency distribution of the stand ages (year last cut) of the southern districts Cherokee National Forest, 1992.

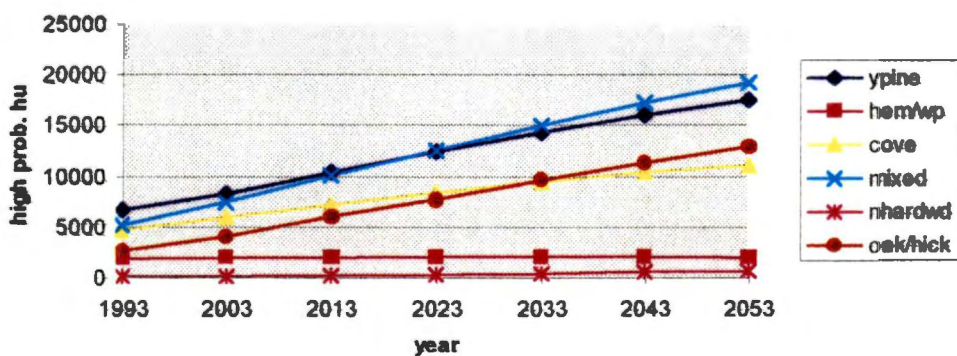
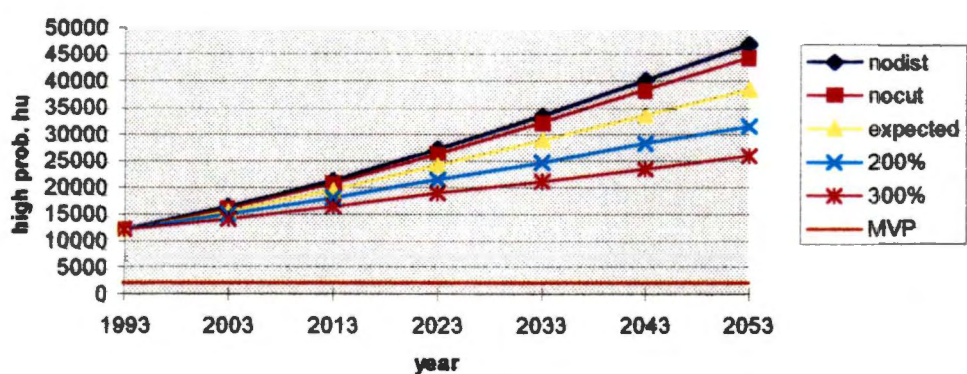
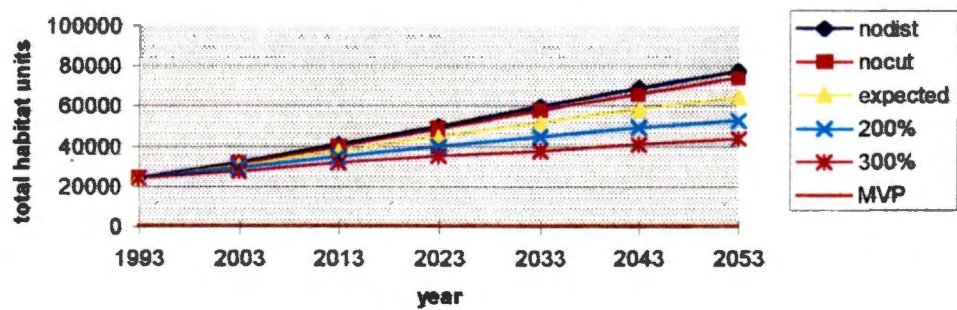


Figure 3. Total habitat units, quality habitat units, and quality habitat units by forest type at expected harvest intensity, 1993-2053 for Acadian flycatcher, southern districts, Cherokee National Forest, Tennessee.

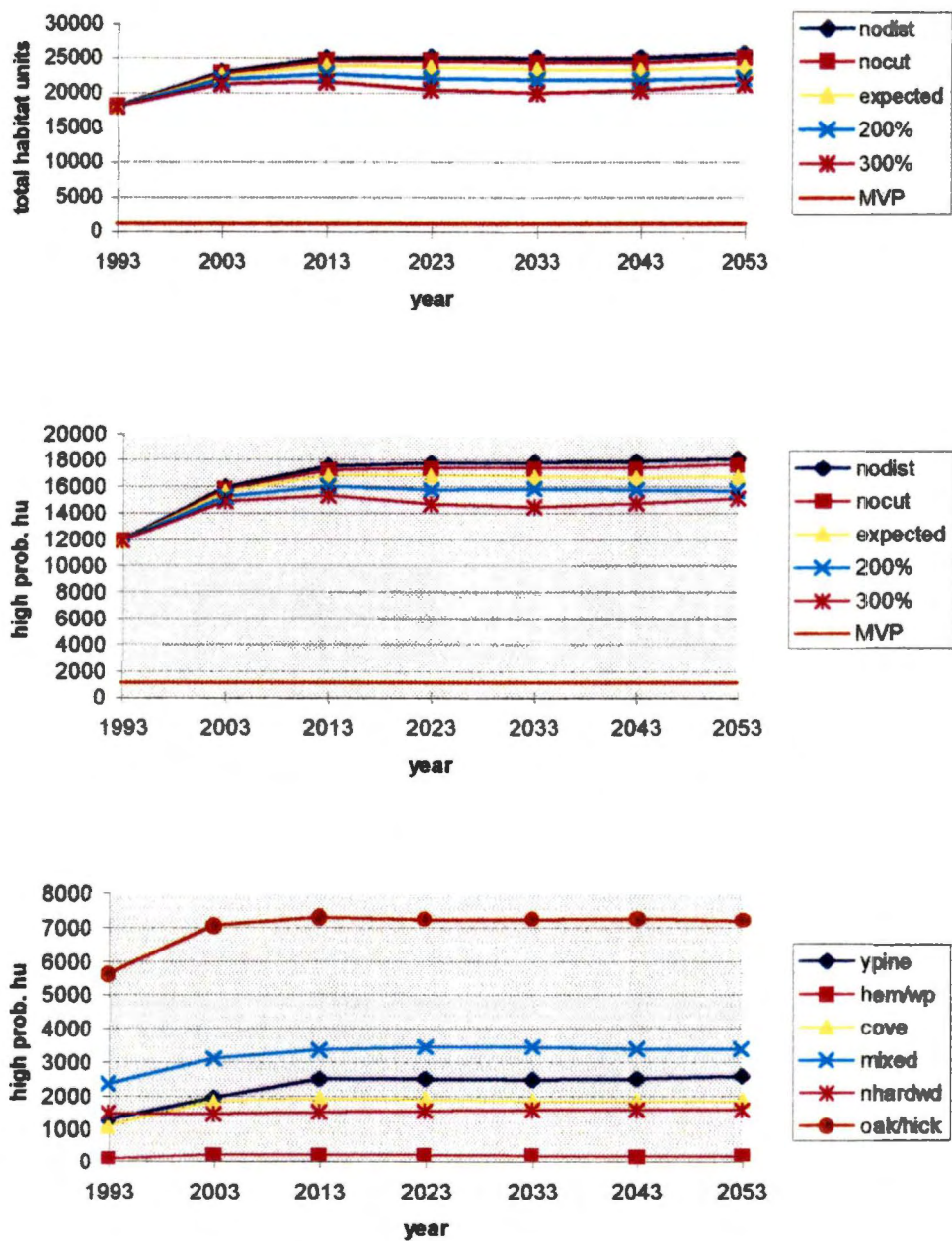


Figure 4. Total habitat units, quality habitat units, and quality habitat units by forest type at expected harvest intensity, 1993-2053 for blue-headed vireo, southern districts, Cherokee National Forest, Tennessee.

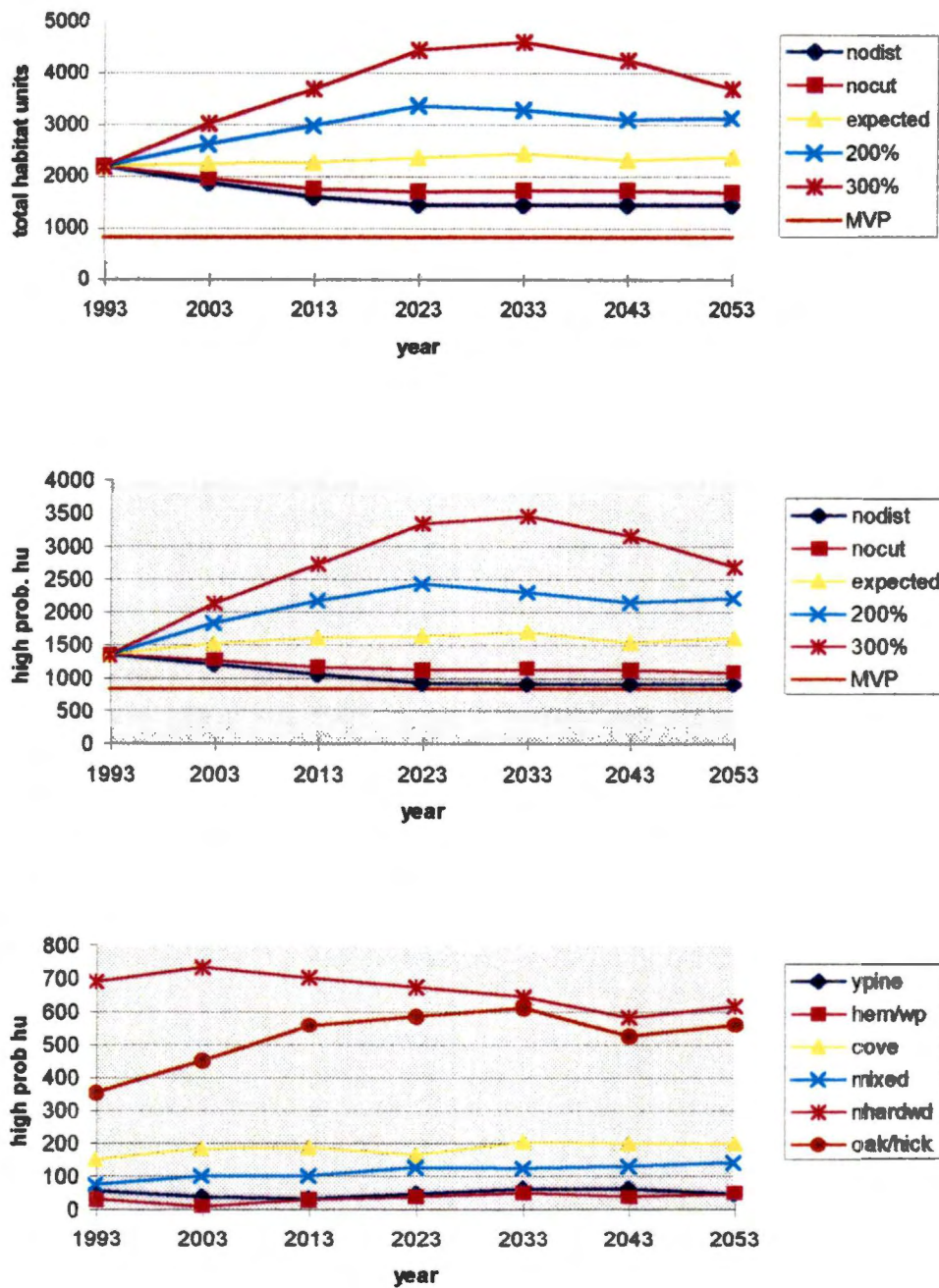


Figure 5. Total habitat units, quality habitat units, and quality habitat units by forest type at expected harvest intensity, 1993-2053 for chestnut-sided warbler, southern districts, Cherokee National Forest, Tennessee.

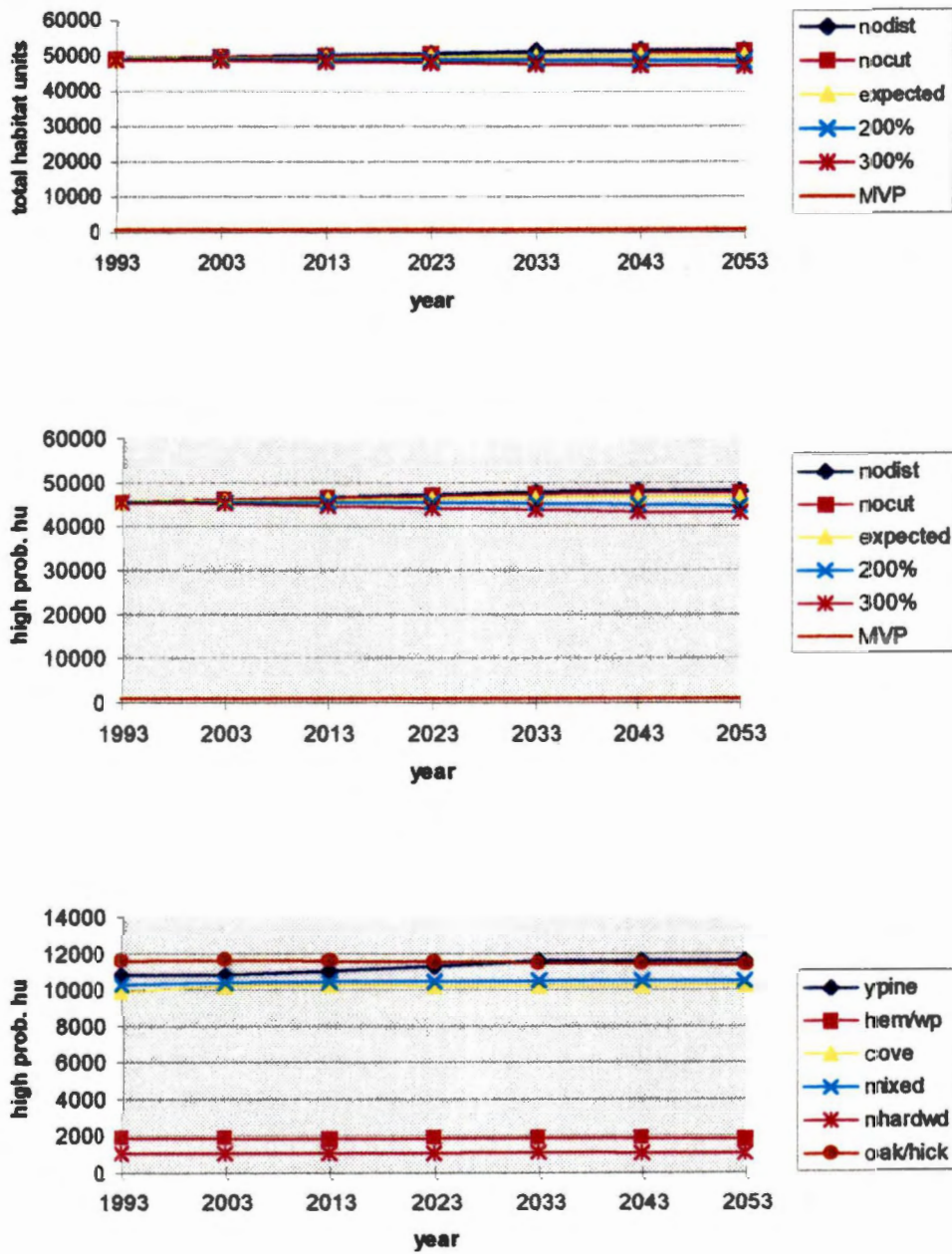


Figure 6. Total habitat units, quality habitat units, and quality habitat units by forest type at expected harvest intensity, 1993-2053 for tufted titmouse, southern districts, Cherokee National Forest, Tennessee.

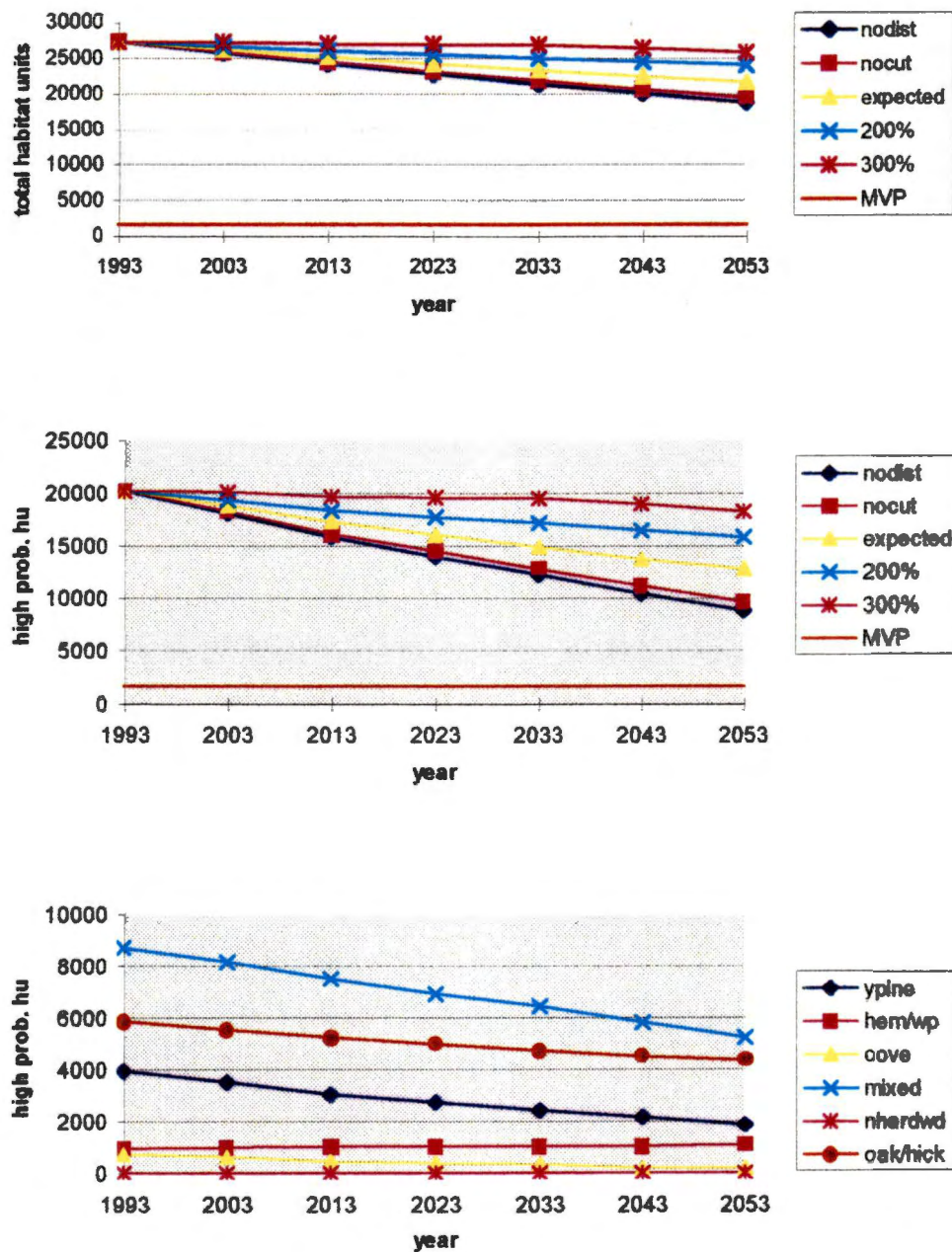


Figure 7. Total habitat units, quality habitat units, and quality habitat units by forest type at expected harvest intensity, 1993-2053 for yellow-billed cuckoo, southern districts, Cherokee National Forest, Tennessee.

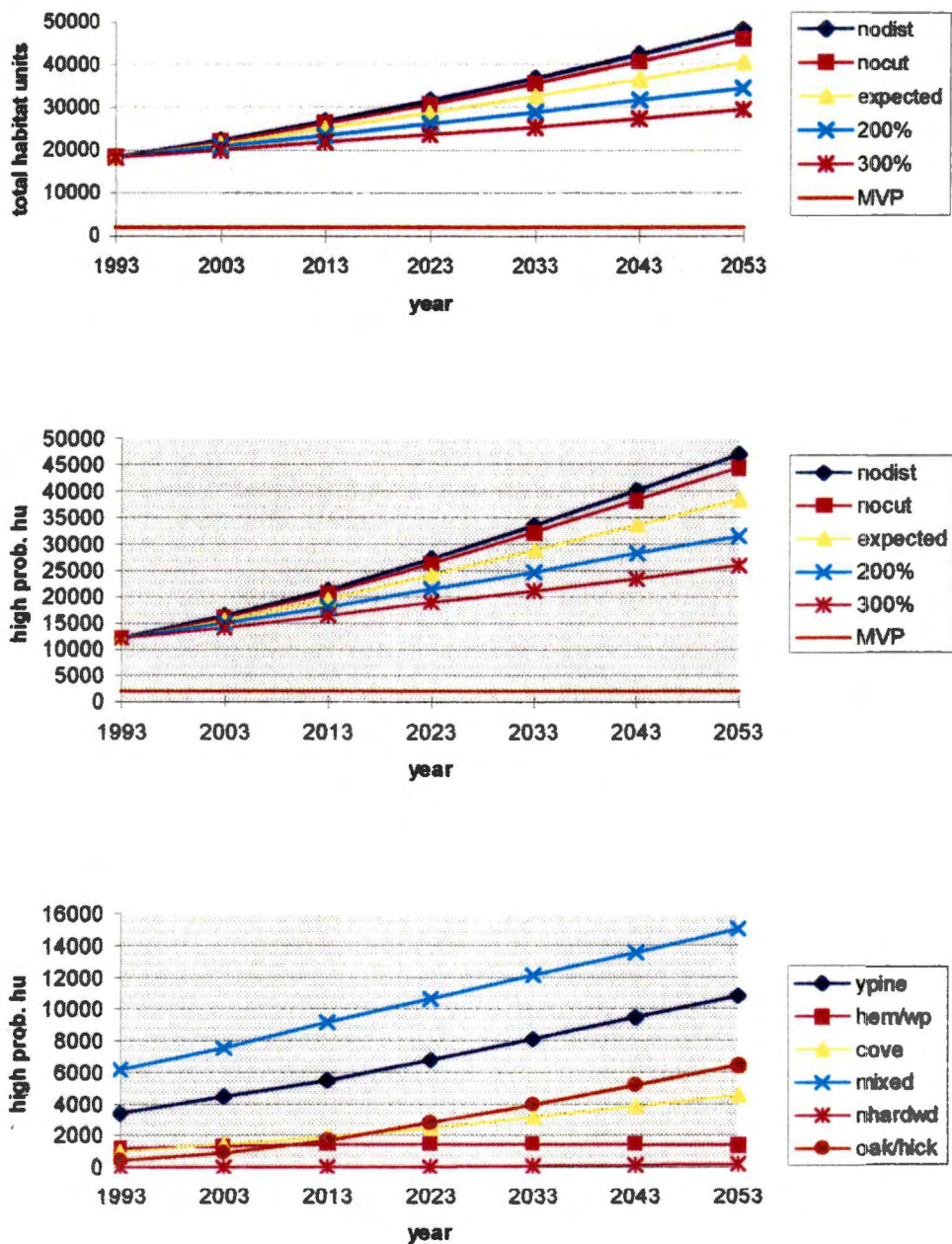


Figure 8. Total habitat units, quality habitat units, and quality habitat units by forest type at expected harvest intensity, 1993-2053 for yellow-throated warbler, southern districts, Cherokee National Forest, Tennessee.

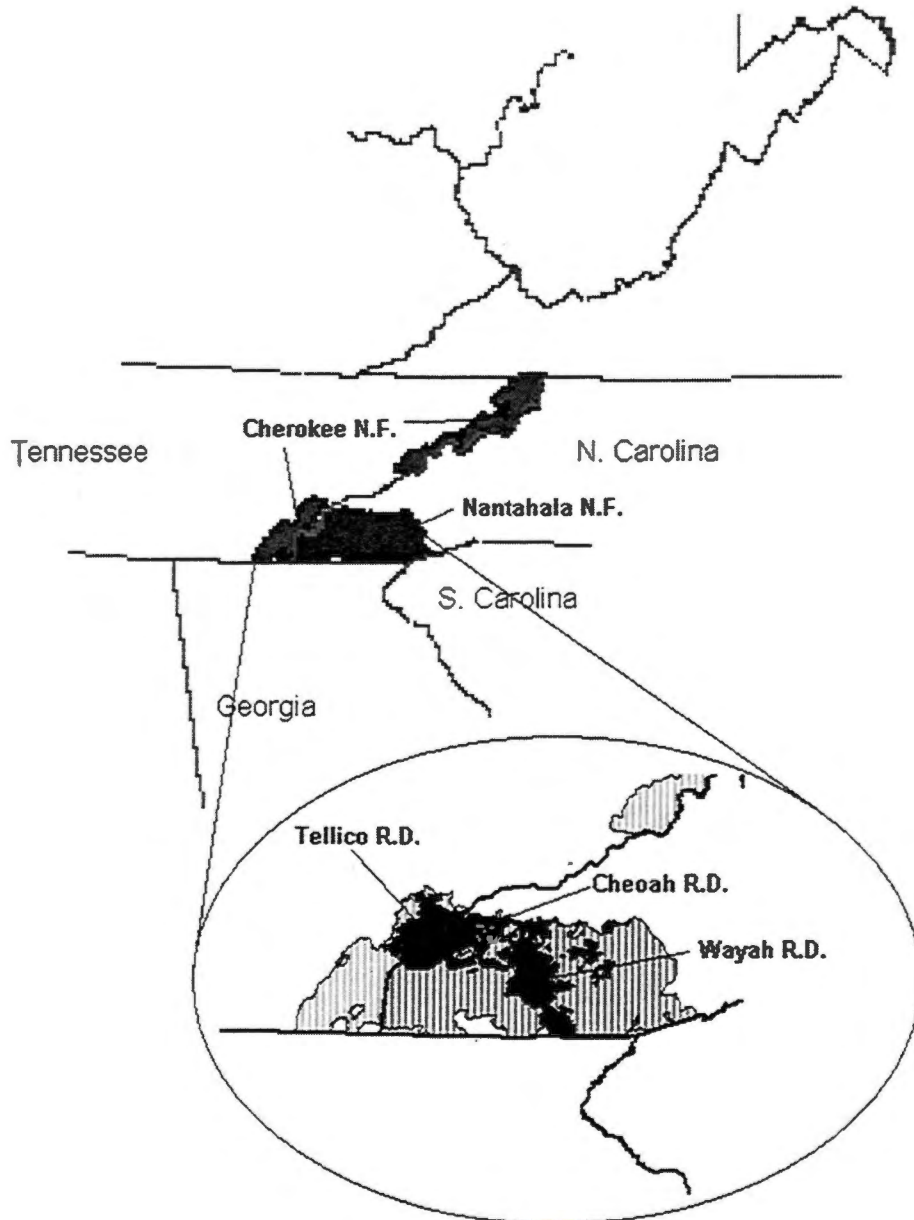


Figure 9. Southern districts, Cherokee National Forest, Tennessee, and the Nantahala National Forest, North Carolina. Our study area is shown in the lower image in black, consisting of the Tellico Ranger District in Cherokee National Forest, and the Cheoah, and Wayah Ranger Districts, Nantahala National Forest.

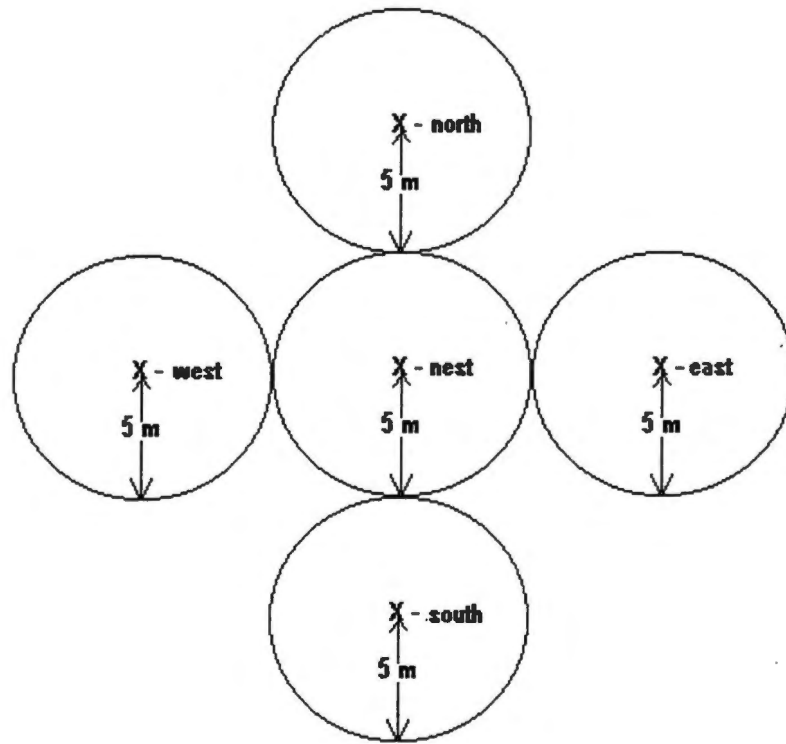


Figure 10. A diagram of the habitat sampling scheme around nest sites. X indicates the center of each plot. The nest plot is centered on the nest itself, the four corollary plot centers are 10 m from the nest in the cardinal directions. Vegetation was sampled within a 5 m radius of plot center for each plot.

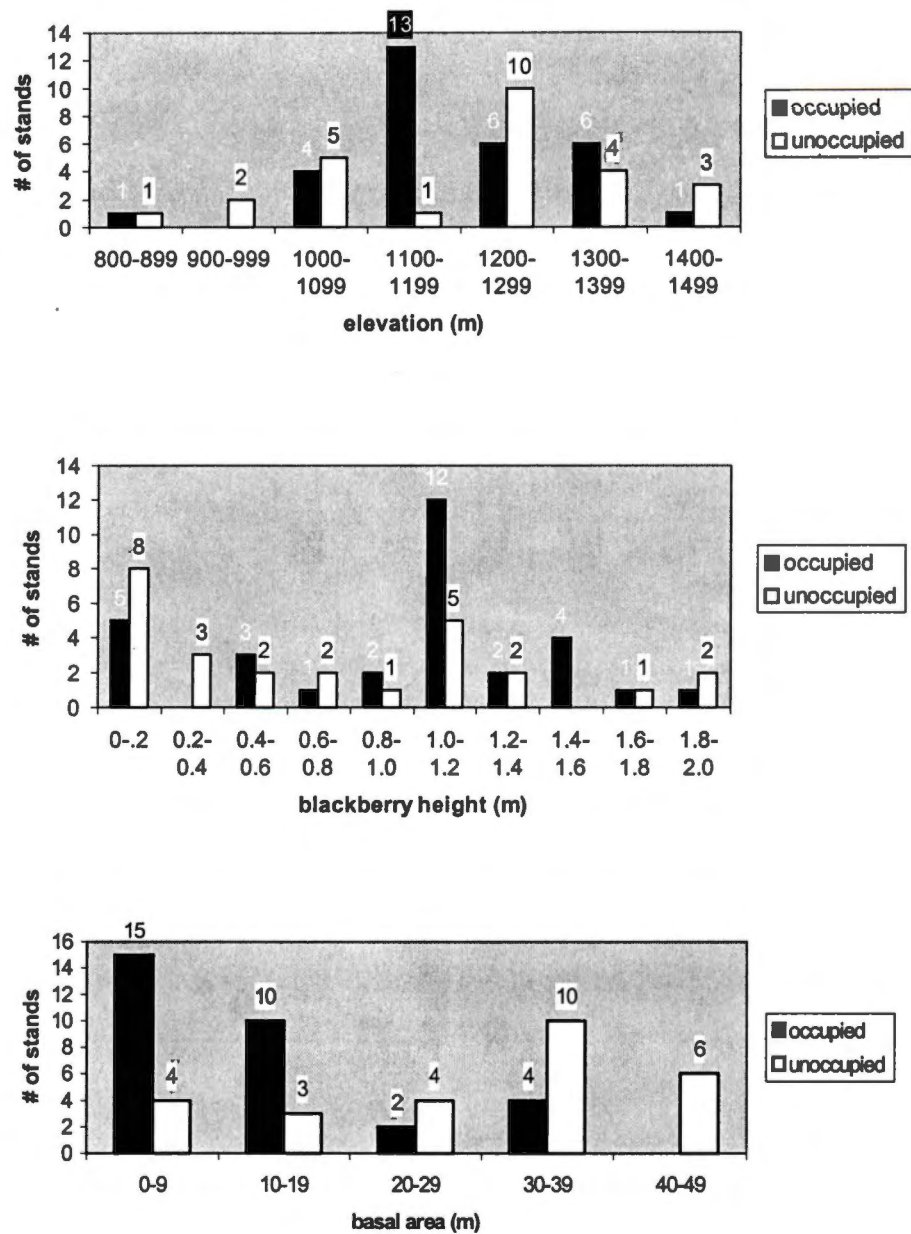


Figure 11. Number of plots where golden-winged warbler was present and absent in 1998 across eleven habitat variables, Cherokee National Forest, Tennessee, and Nantahala National Forest, North Carolina.

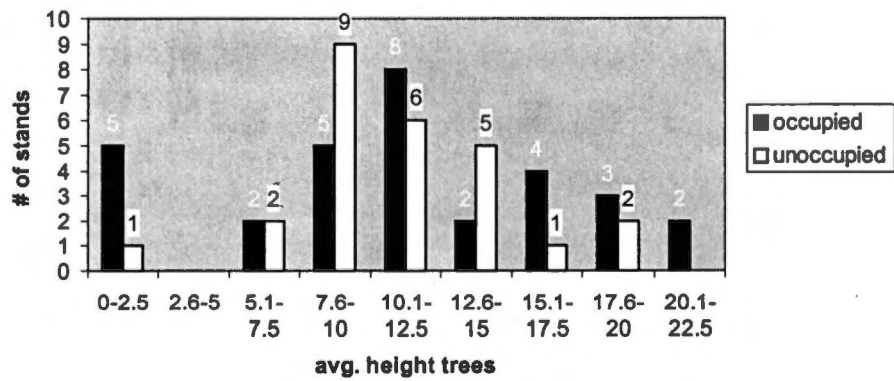
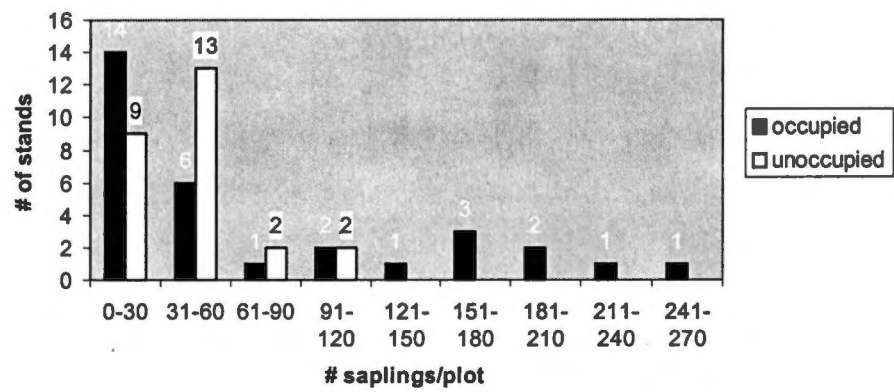
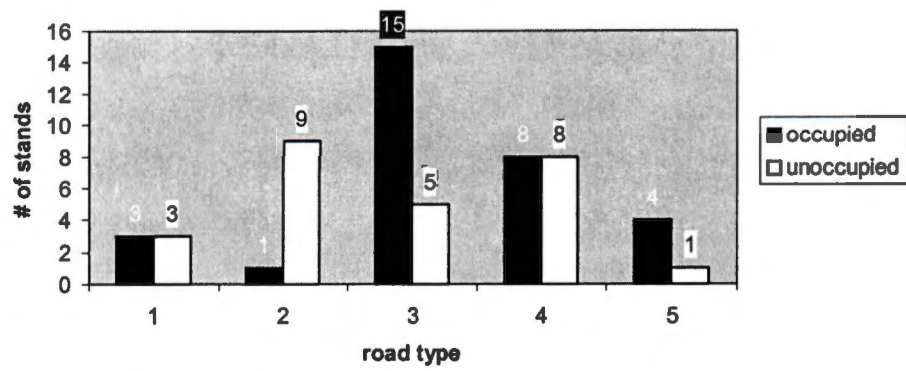


Figure 11. (continued)

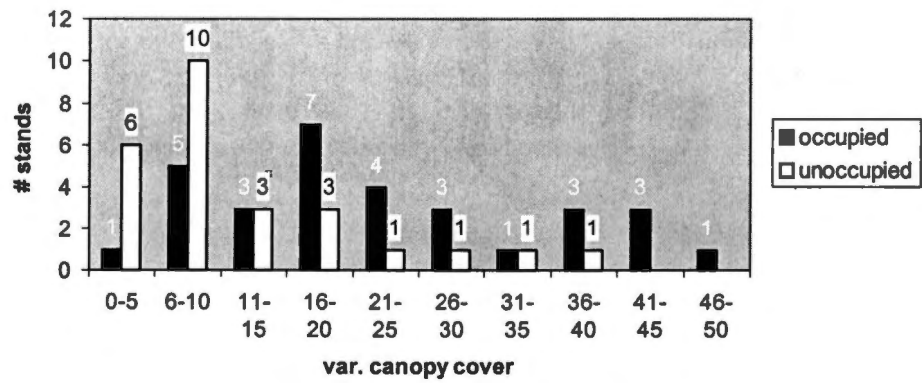
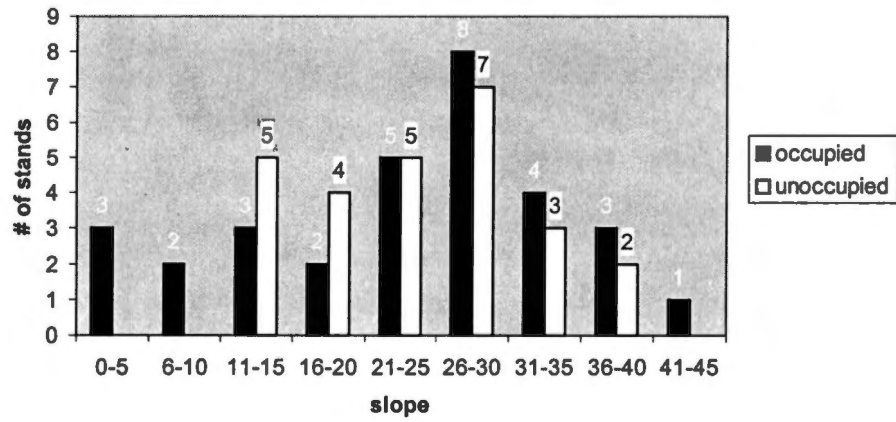
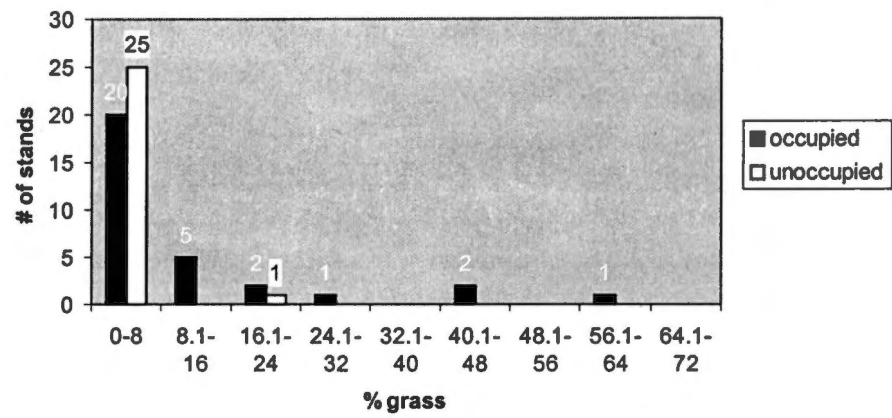


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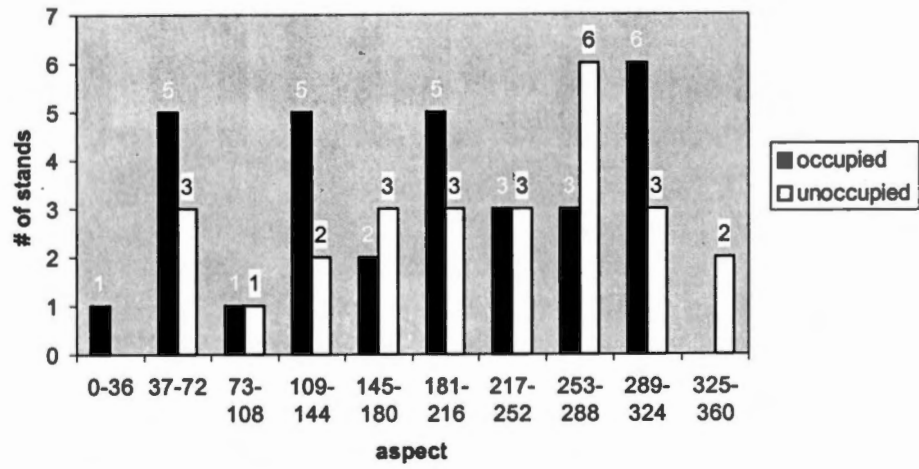
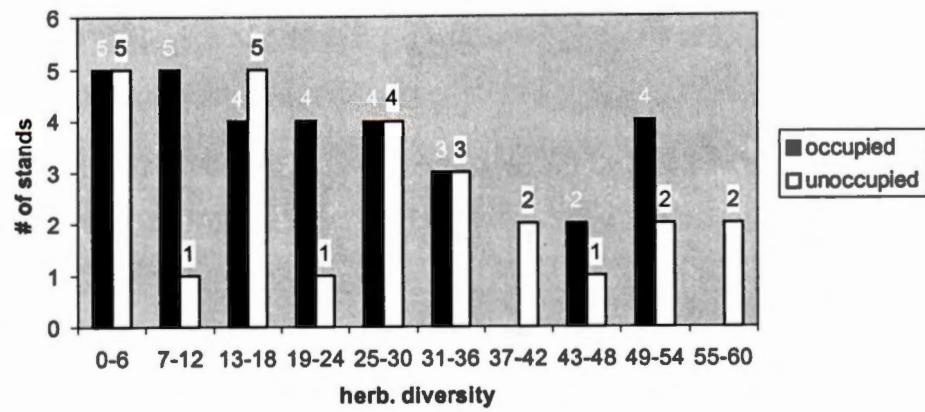


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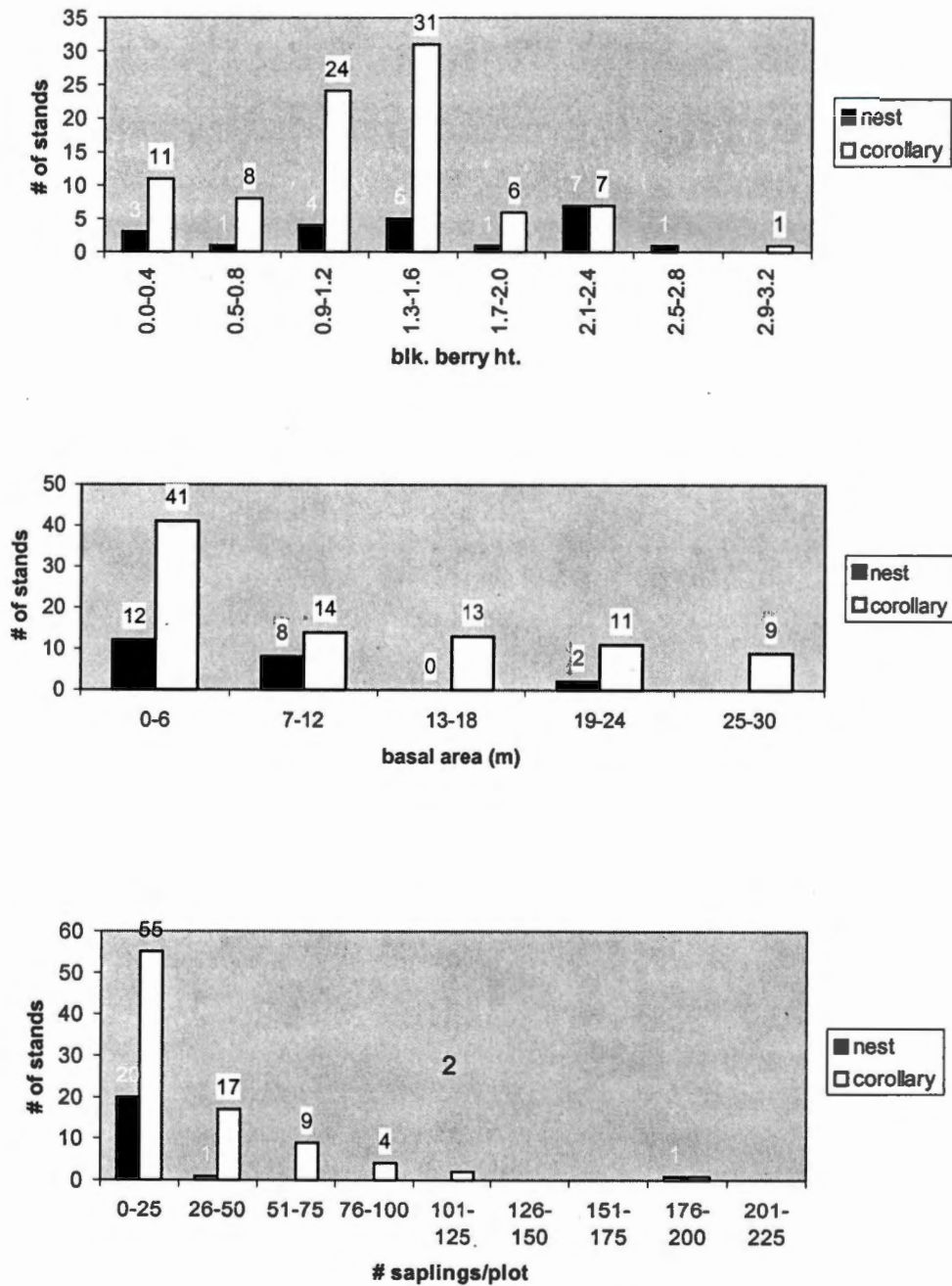


Figure 12. Number of golden-winged warbler nests and surrounding plots in 1998 across eight habitat variables, Cherokee National Forest, Tennessee, and Nantahala National Forest, North Carolina.

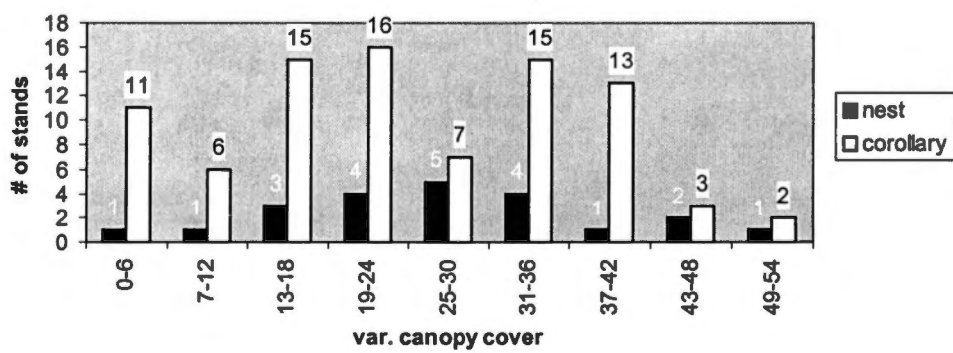
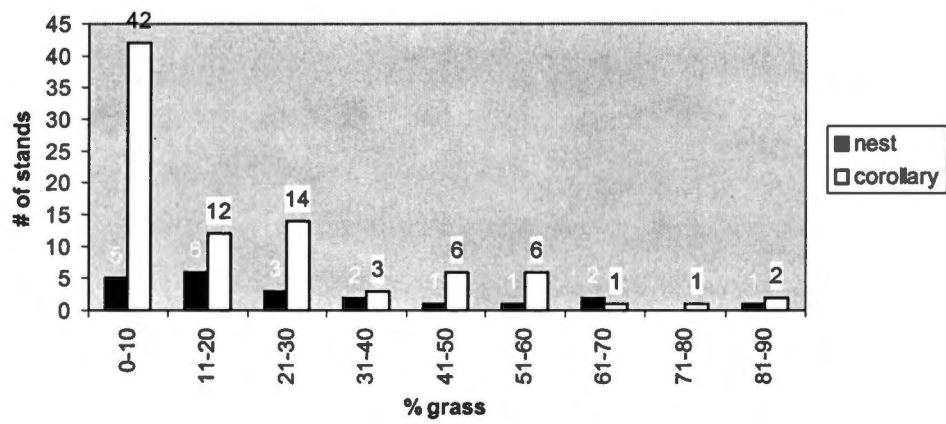
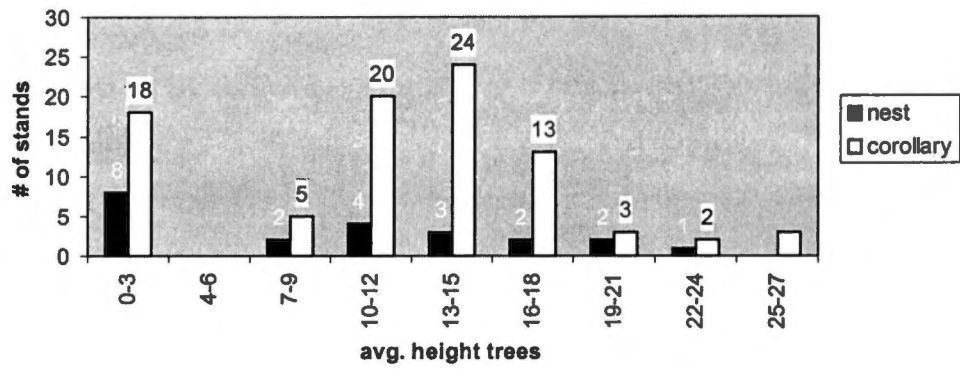


Figure 12. (continued)

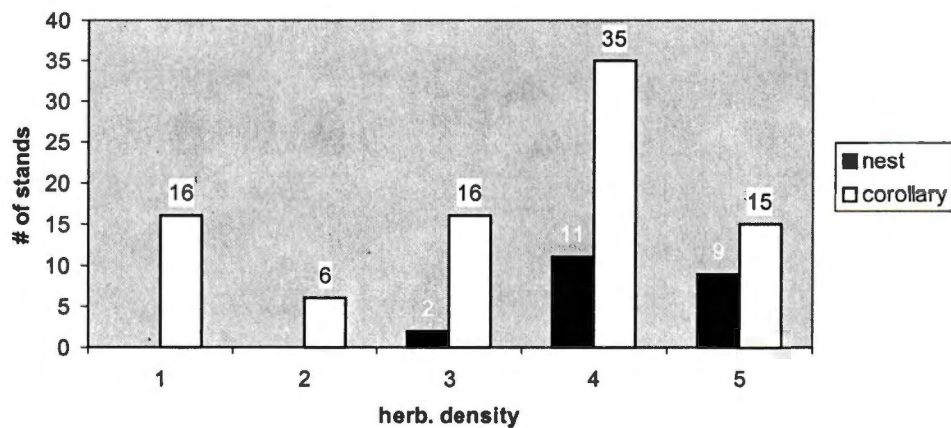
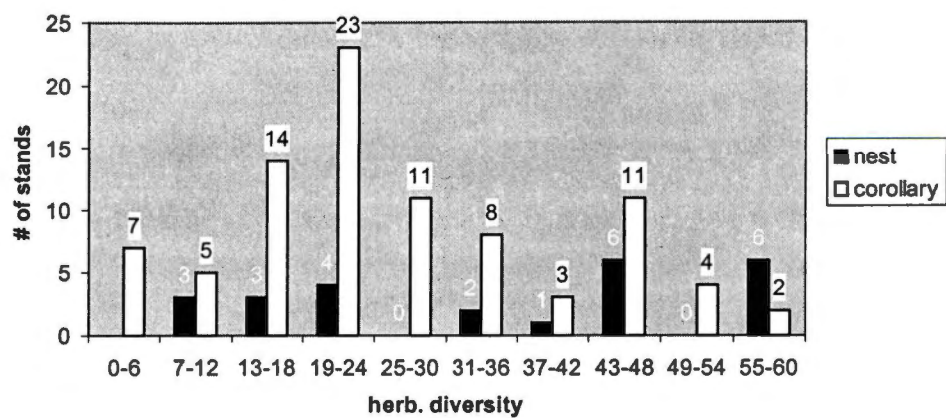


Figure 12. (continued)

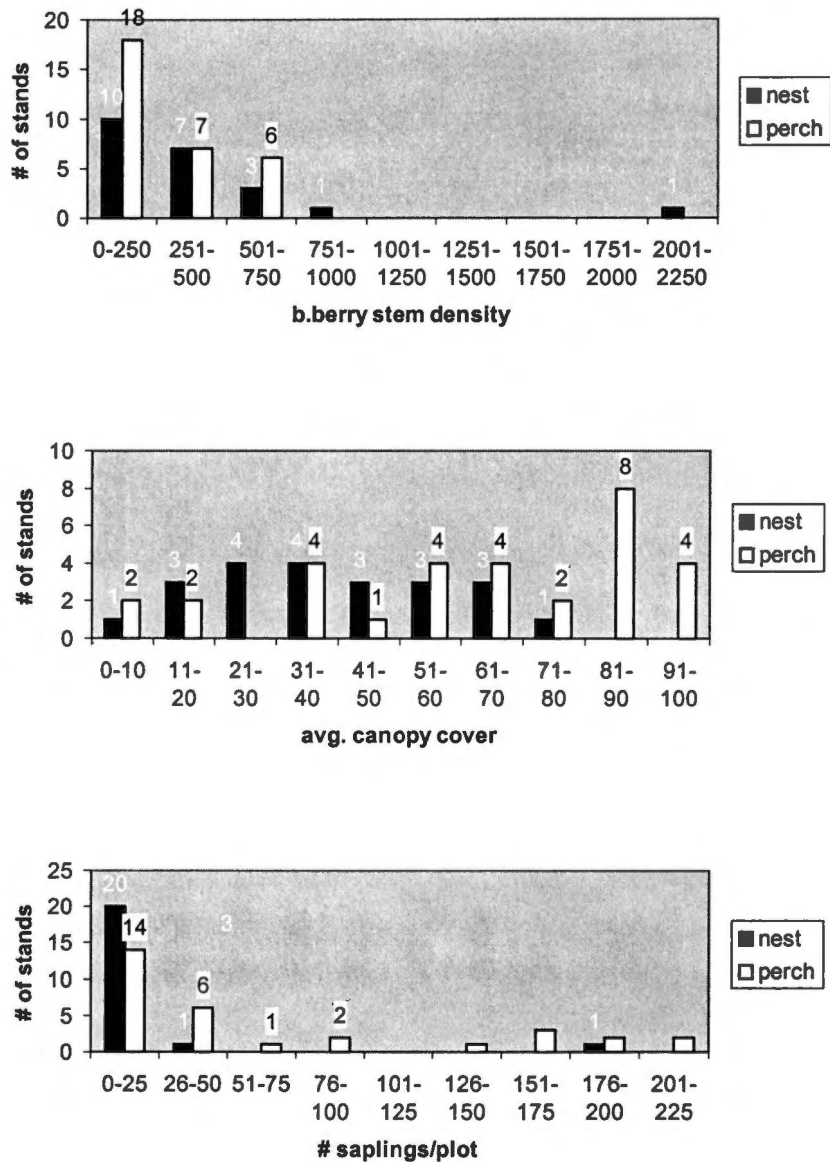


Figure 13. Number of plots with golden-winged warbler territories and nests across four habitat variables in 1998, Cherokee National Forest, Tennessee, and Nantahala National Forest, North Carolina.

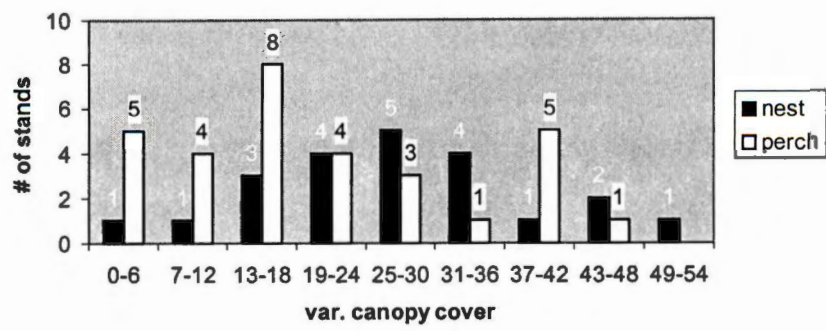


Figure 13. (Continued)

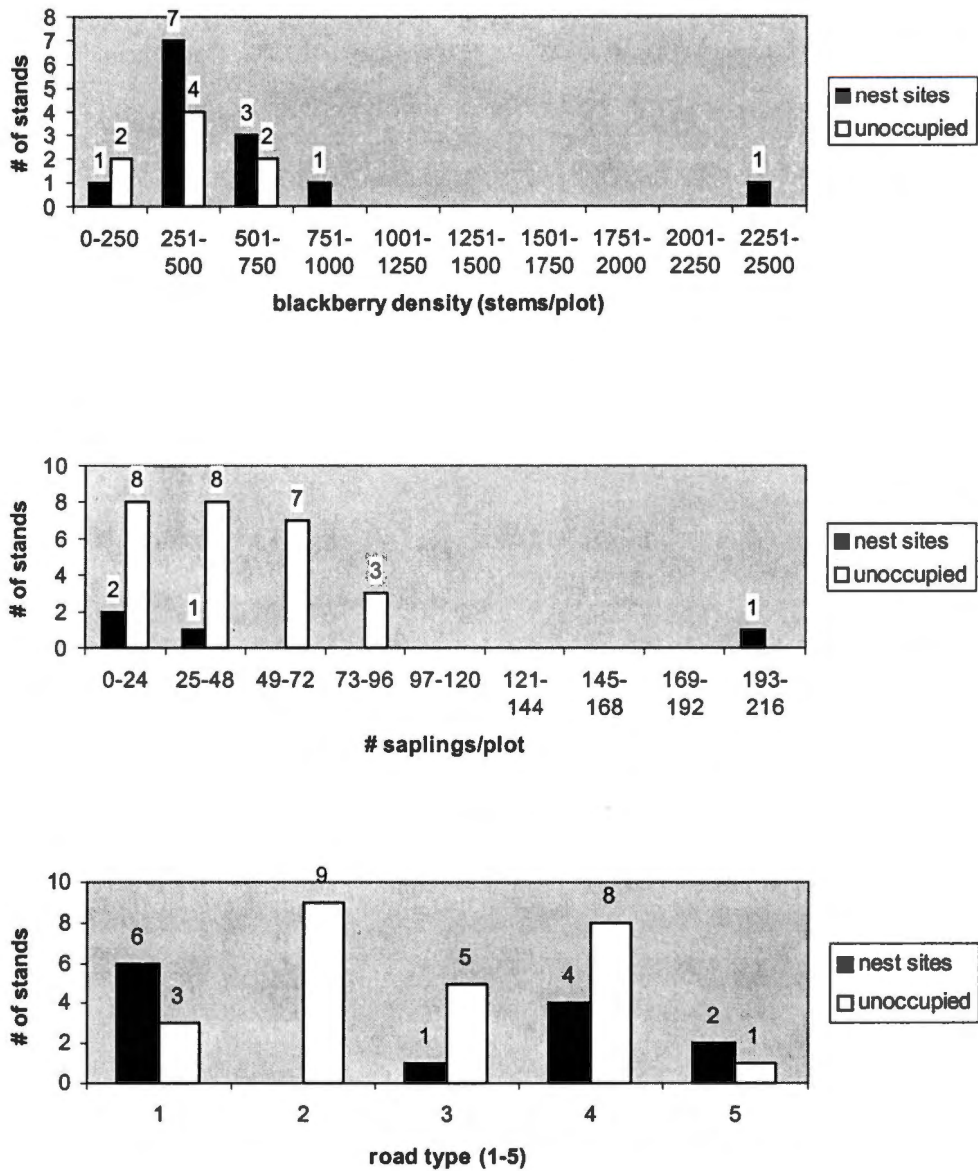


Figure 14. Number of plot with nests and number of unoccupied territories in 1998 across ten habitat variables, Cherokee National Forest, Tennessee, and Nantahala National Forest, North Carolina.

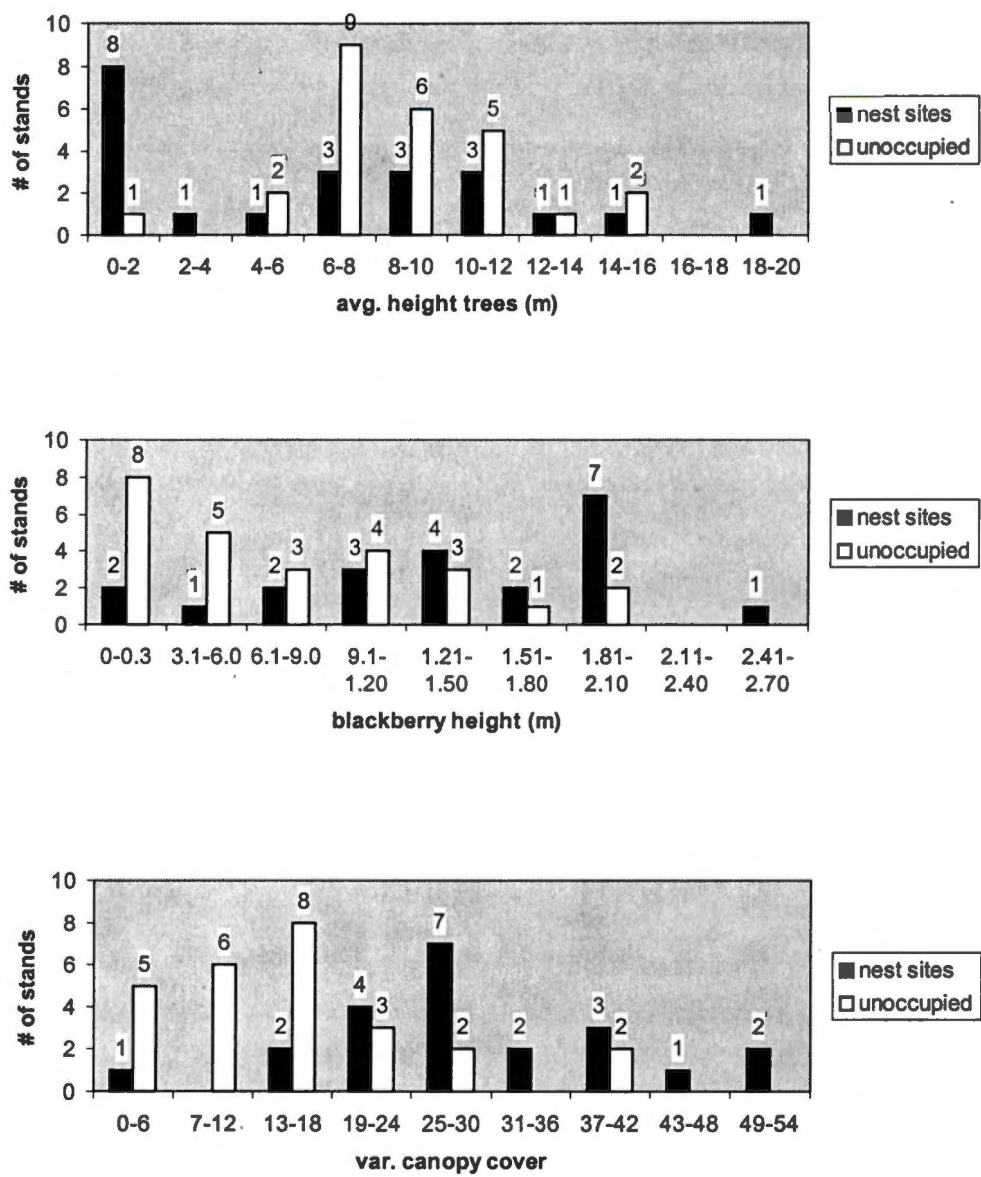


Figure 14. (continued)

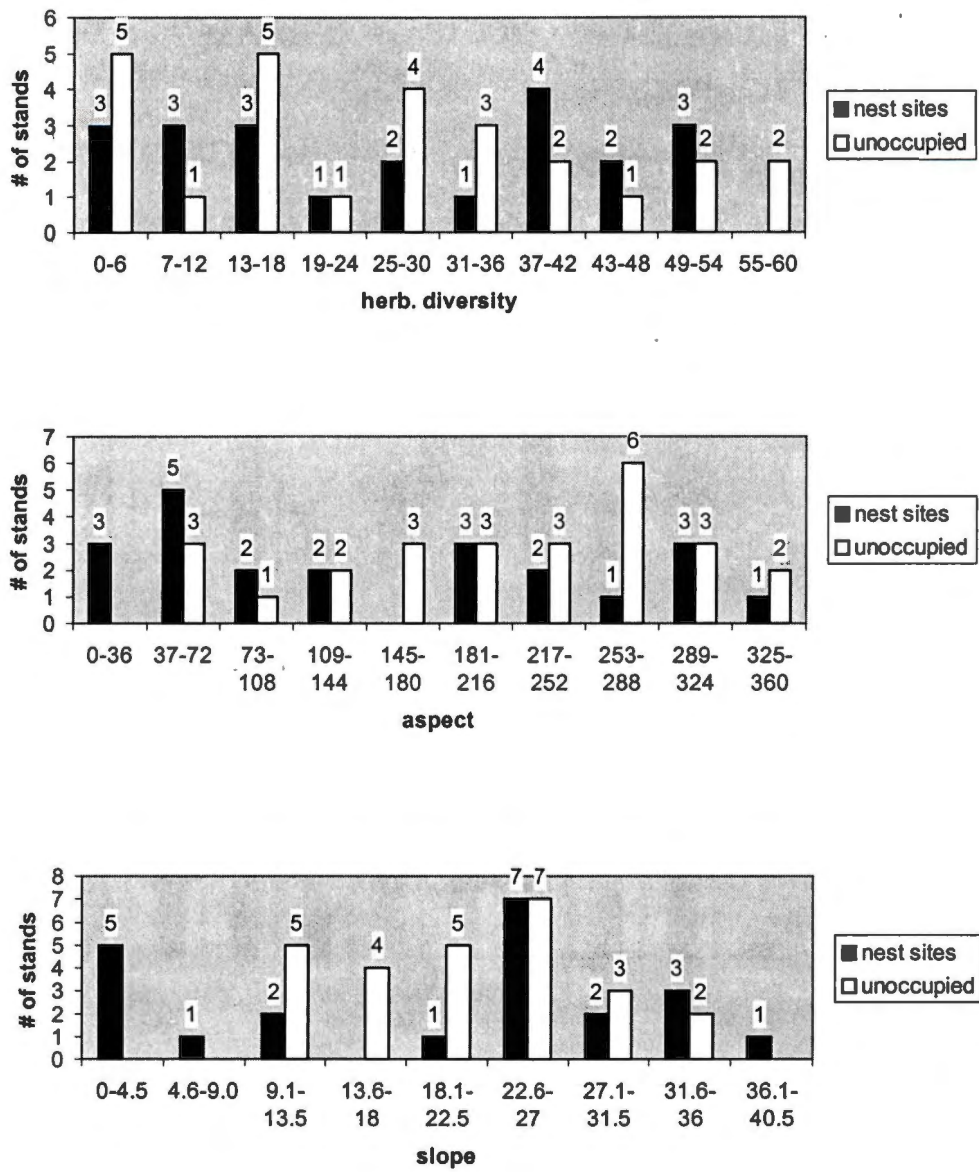


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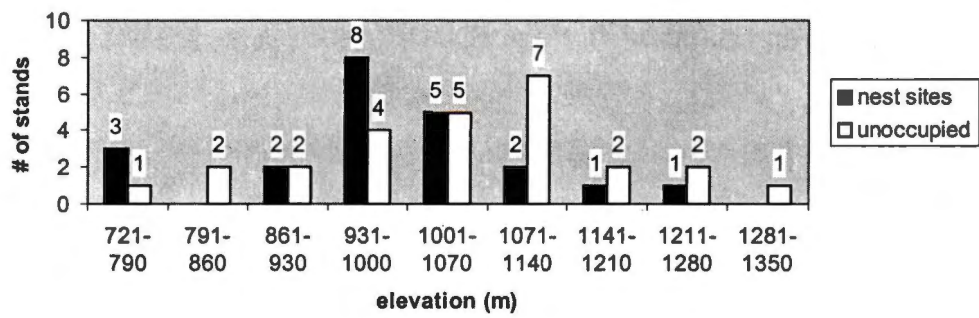


Figure 14. (continued)

VITA

Nathan Klaus was born in Springfield, Illinois in 1972. He lived much of his life in Iowa and Nebraska. He graduated from Abraham Lincoln High School in Council Bluffs Iowa in 1990. It was in the Loess Hills surrounding Council Bluffs that Nathan developed a deep appreciation of the outdoors and a commitment to their protection and preservation. Following this conviction he received his Bachelor of Science in Biology from the University of Iowa in May of 1994.

Nathan's work in the Southern Appalachians has served to deepen his belief in conservation while also educating him in the complexities of land management, natural disturbance, and multiple use. Nathan received his Master of Science degree in Wildlife and Fisheries Science from the University of Tennessee, Knoxville, in August 1999. Nathan has accepted a position with the Georgia Department of Natural Resources, Non-game and Endangered Wildlife Program, where he works with non-game birds in the Georgia Piedmont.

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