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# Multiscale Habitat Suitability Index Models for Priority Landbirds in the Central Hardwoods and West Gulf Coastal Plain/Ouachitas Bird Conservation Regions

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## Abstract

Ecoregional conservation planning for priority landbirds requires methods that explicitly link populations to habitat conditions at multiple scales. We developed Habitat Suitability Index (HSI) models to assess habitat quality for 40 priority bird species in the Central Hardwoods and West Gulf Coastal Plain/Ouachitas Bird Conservation Regions. The models incorporated both site and landscape environmental variables derived from one of six nationally consistent datasets: ecological subsections from the National Ecological Unit Hierarchy, National Land Cover Dataset, National Elevation Dataset, National Hydrography Dataset, State Soil Geographic Database, and Forest Inventory and Analysis data. We initially defined potential habitat for each species from unique landform, landcover, and successional age class combinations. Species-specific environmental variables identified from the literature were used to refine initial habitat estimates. We verified models by comparing subsection-level HSI scores and Breeding Bird Survey (BBS) abundance via Spearman rank correlation. To validate models, we developed generalized linear models that predicted BBS abundance as a function of HSI score and Bird Conservation Region. We considered models that included a significant ($P \leq 0.100$) positive coefficient on the BBS predictor to be valid and useful for conservation planning.

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## Cover Photos

Clockwise from top left:
Cerulean warbler, U.S. Forest Service; Wood thrush, Steve Maslowski, U.S. Fish & Wildlife Service; Pileated woodpecker, U.S Forest Service; Painted bunting, Deanna K. Dawson, Patuxent Bird Identification InfoCenter, Photo used with permission; Kentucky warbler, U.S. Fish & Wildlife Service; Bewick's wren, Dave Menke, U.S. Fish & Wildlife Service.

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# INTRODUCTION

The primary goal of the North American Landbird Conservation Plan (Rich and others 2004) is to create landscapes that can sustain populations of the 448 native landbird species that breed in the United States and Canada. To attain this goal, the Plan advocates a three-phase approach:

1. Establish population objectives at the continental scale.
2. Allocate these population objectives to specific Bird Conservation Regions (BCRs).
3. Translate the regional population objectives to habitat goals within each BCR.

The first two steps of this process have been completed (Panjabi and others 2001, Rosenberg and Blancher 2005), and it is at this third step where the conservation community stands today.

Translating target population numbers into concrete habitat goals requires both knowledge of how landbird populations respond to changing habitat conditions and a method for quantifying this relationship. However, there are few data explicitly linking landbird abundance to specific habitat conditions, nor is there consensus on the optimal methodology to achieve this linkage. The goal of our research is to develop a comprehensive, replicable approach to ecoregional habitat assessment that links habitat conditions to the density of priority bird species. Specific objectives are to:

1. Assess the ability of landscapes to sustain priority species at prescribed population levels based on the extent and distribution of available habitats.
2. Monitor changes in the ability of landscapes to sustain species.
3. Predict how landscape suitability changes under alternative succession and disturbance patterns, land use, conservation strategies, management practices, and development pressures.

To create a replicable and transferable methodology, we selected a Habitat Suitability Index (HSI) modeling approach. HSI models were initially developed by the U.S. Department of the Interior (USDI) Fish and Wildlife Service (FWS) to evaluate habitat quality for a variety of species (Schamberger and others 1982). These models identify and quantify the relationship between key environmental variables and habitat suitability on a scale from 0 to 1. HSI scores are calculated independently for each environmental factor and an appropriate weighting scheme is used to combine individual variables and determine a composite suitability index (SI) score for a particular location. Although the FWS developed HSI models solely with site-specific habitat variables (e.g., canopy cover) for assessing stand-level habitat suitability, researchers are increasingly developing HSI models that incorporate broad-scale metrics (e.g., percent forest in a 1-km radius) for application to large landscapes (Larson and others 2003). The continued use of the HSI approach by both researchers and managers likely is a result of the intuitive nature of these models as well as their scalability and portability to novel situations. HSI models easily incorporate existing information via a priori hypotheses but also allow generalization of habitat relationships across areas and species where empirical data are limited. Currently, few HSI models include environmental variables at both the site and landscape scale due to the limited site-specific data across areas that are large enough to exhibit strong differences in landscape structure or composition. Nevertheless, habitat selection by birds is a multiscale process (Villard and others 1998) and habitat models should reflect conditions at multiple scales. This report begins filling this gap by documenting multiscale HSI models for 40 priority landbird species (Table 1).

**Table 1.—Partners in Flight regional combined score and USDI Fish and Wildlife Service Bird of Conservation Concern status for 40 priority landbird species in the Central Hardwoods and West Gulf Coastal Plain/Ouachitas Bird Conservation Regions**

| | | Central Hardwoods | | West Gulf Coastal Plain/Ouachitas | |
|---|---|---|---|---|---|
| Species | Alpha code[a] | Regional combined score | Bird of Conservation Concern | Regional combined score | Bird of Conservation Concern |
| Acadian flycatcher | ACFL | 16 | No | 17 | Yes |
| American woodcock | AMWO | -- | No | -- | No |
| Bachman's sparrow | BACS | 20 | Yes | 20 | Yes |
| Bell's vireo | BEVI | 15 | Yes | 16 | Yes |
| Bewick's wren | BEWR | 15 | Yes | 16 | Yes |
| Black-and-white warbler | BAWW | 13 | No | 16 | No |
| Blue-gray gnatcatcher | BGGN | 14 | No | 13 | No |
| Blue-winged warbler | BWWA | 19 | Yes | -- | No |
| Brown thrasher | BRTH | 15 | No | 13 | No |
| Brown-headed nuthatch | BHNU | 19 | No | 19 | Yes |
| Carolina chickadee | CACH | 15 | No | 16 | No |
| Cerulean warbler | CERW | 19 | Yes | 19 | Yes |
| Chimney swift | CHSW | 16 | No | 14 | No |
| Chuck-will's-widow | CWWI | 14 | No | 16 | Yes |
| Eastern wood-pewee | EAWP | 15 | No | 16 | No |
| Field sparrow | FISP | 17 | No | 15 | No |
| Great crested flycatcher | GCFL | 13 | No | 13 | No |
| Hooded warbler | HOWA | 13 | No | 16 | No |
| Kentucky warbler | KEWA | 18 | No | 19 | Yes |
| Louisiana waterthrush | LOWA | 15 | Yes | 18 | Yes |
| Mississippi kite | MIKI | 14 | No | 16 | No |
| Northern bobwhite | NOBO | 16 | No | 15 | No |
| Northern parula | NOPA | 12 | No | 13 | No |
| Orchard oriole | OROR | 17 | No | 18 | Yes |
| Painted bunting | PABU | 16 | No | 17 | No |
| Pileated woodpecker | PIWO | 13 | No | 16 | No |
| Prairie warbler | PRAW | 18 | Yes | 18 | Yes |
| Prothonotary warbler | PROW | 14 | No | 17 | Yes |
| Red-cockaded woodpecker | RCWO | 21 | No | 21 | No |
| Red-headed woodpecker | RHWO | 16 | Yes | 17 | Yes |
| Swainson's warbler | SWWA | 20 | Yes | 20 | Yes |
| Swallow-tailed kite | STKI | 19 | No | 18 | Yes |
| Whip-poor-will | WPWI | 17 | Yes | 13 | No |
| White-eyed vireo | WEVI | 15 | No | 16 | No |
| Wood thrush | WOTH | 16 | Yes | 15 | Yes |
| Worm-eating warbler | WEWA | 18 | Yes | 15 | Yes |
| Yellow-billed cuckoo | YBCU | 13 | No | 15 | No |
| Yellow-breasted chat | YBCH | 16 | No | 13 | No |
| Yellow-throated vireo | YTVI | 16 | No | 15 | No |
| Yellow-throated warbler | YTWA | 15 | No | 16 | No |

[a] Pyle and DeSante (2003).

## STUDY AREAS

We developed HSI models for landbirds identified as priorities in the Central Hardwoods (CH) and West Gulf Coastal Plain/Ouachitas (WGCP) BCRs (Fig. 1). The CH, approximately 33 million ha straddling the Mississippi River, is dominated by deciduous hardwood forest. This region is bordered to the north and west by the tallgrass prairie ecosystem, to the east by the Appalachian Mountains, and to the south by the southern pine belt along the Coastal Plain. The vast forests of the CH make it an important breeding area for many area-sensitive species,

Figure 1.—Central Hardwoods and West Gulf Coastal Plain/Ouachitas Bird Conservation Regions.

including the cerulean warbler, Kentucky warbler, Louisiana waterthrush, and worm-eating warbler (Panjabi and others 2001). The WGCP also is predominantly forested but consists primarily of pine: longleaf pine in the south transitioning to loblolly and shortleaf pine in the north. As a result, this region contains large populations of pine specialists (e.g., red-cockaded woodpecker, brown-headed nuthatch, and pine warbler). The WGCP also contains broad swaths of bottomland hardwood forest, particularly along the Arkansas, Ouachita, and Sabine Rivers, which support substantial populations of the hooded warbler, Kentucky warbler, and Swainson's warbler (Conner and Dickson 1997).

# METHODS

## Priority Bird Species

We selected priority bird species for modeling by identifying a subset of the forest-breeding landbirds in the CH or WGCP with a Partners in Flight (PIF) regional combined score of at least 15 (Panjabi and others 2005) or an FWS designation as a Bird of Conservation Concern (USDI Fish and Wildl. Serv. 2002) (Table 1). Forty-nine species initially met these criteria. We eliminated Bachman's warbler and the ivory-billed woodpecker from consideration due to limited habitat and validation data available within the CH and WGCP for these species. Also, we did not model habitat suitability for the ruffed grouse, broad-winged hawk, eastern kingbird, scissor-tailed flycatcher, loggerhead shrike, summer tanager, or eastern towhee. We added American woodcock, blue-gray gnatcatcher, great crested flycatcher, and northern parula to ensure the species modeled were representative of a cross section of habitat associations (e.g., early successional forest, pine savanna, bottomland hardwoods) and conservation priorities (e.g., critical recovery, management attention, planning and responsibility) within these BCRs.

## HSI Model Development

In our adaptation of the HSI approach, we assume that habitat suitability is a function of both composition and structure at the site and landscape scales. To characterize environmental variables at each of these scales, we relied on six nationally consistent datasets:

1. Ecological subsections from the National Ecological Unit Hierarchy.
2. National Landcover Dataset (NLCD) (30-m pixels).
3. National Elevation Dataset (NED) (30-m pixels).
4. National Hydrography Dataset (NHD).
5. State Soil Geographic Database (STATSGO).
6. Forest Inventory and Analysis (FIA) data.

The first five datasets are widely available and commonly used to characterize landscape composition and structure. The sixth, FIA, provides information on the composition and structure of vegetation within forest patches (i.e., site scale) from a national field survey of forest lands undertaken by the USDA Forest Service. A description of the methodology used to integrate these datasets in a spatially explicit framework is available in Tirpak and others (2009b).

**Table 2.—Parameters and data sources for inputs in priority forest-breeding landbird Habitat Suitability Index models, Central Hardwoods and West Gulf Coastal Plain/Ouachitas Bird Conservation Regions; numbers correspond to Suitability Index (SI) functions in text**

| | Species code[a] | | | | | |
|---|---|---|---|---|---|---|
| Data source | ACFL | AMWO | BACS | BEVI | BEWR | BAWW |
| DEM, NLCD, and FIA | | | | | | |
| Landform, landcover, and successional age class | 1 | 1 | 1 | 1 | 1 | 1 |
| NLCD and FIA | | | | | | |
| Early successional patch size (ha) | | | | | | |
| NLCD and NHD | | | | | | |
| Occurrence of water | | | | | | |
| Distance (m) to water | 2 | | | | | |
| NLCD | | | | | | |
| Forest patch size (ha) | 4 | | 2 | | | 2 |
| Landscape composition (percent forest in 1-km radius) | 5 | | | | | 3 |
| Landscape composition (percent forest in 10-km radius) | | | | | | |
| Occurrence of edge | | | | 3 | | |
| Distance (m) to edge | | | | | | |
| Interspersion – 1 landcover class | | | | | | |
| Interspersion – 2 landcover classes | | 3 | | 2 | 2 | |
| Connectivity (km) | | | 4 | | | |
| Grass-open landcover | | | | | | |
| FIA | | | | | | |
| Basal area ($m^2$/ha) | | | | | | |
| Hardwood basal area ($m^2$/ha) | | | | | | |
| Pine basal area ($m^2$/ha) | | | | | | |
| Sawtimber (> 28 cm d.b.h.) tree density (trees/ha) | | | | | | |
| Large (> 50 cm d.b.h) tree density (trees/ha) | | | | | | |
| Large (> 35 cm d.b.h) pine density (trees/ha) | | | | | | |
| Dominant (> 76.2 cm d.b.h.) tree density (trees/ha) | | | | | | |
| Midstory (11–25 cm d.b.h.) density (trees/ha) | | | | | | |
| Snag density (snags/ha) | | | | | 3 | |
| Large (> 30 cm d.b.h.) snag density (snags/ha) | | | | | | |
| Canopy cover (percent) | 3 | | 3 | | | 4 |
| Small stem (< 2.5 cm d.b.h.) density (stems/ha) | | 2 | | 4 | | |
| DEM | | | | | | |
| Slope | | | | | | |
| NHD | | | | | | |
| Distance (m) to stream | | | | | | |
| STATSGO | | | | | | |
| Soil texture | | 4 | | | | |
| Soil moisture | | 5 | | | | |

**continued**

As a first step in developing HSI models, we identified key habitat factors for each species from the literature and compiled all pertinent data from these sources. In the interests of parsimony and processing time, we generally limited our HSI models to five or fewer suitability indices (Table 2). The first SI in all models (with the exception of chimney swift) was a function that assigned SI scores to unique combinations of landform, landcover, and successional age classes. Landform comprised three classes (floodplain-valley, terrace-mesic, and xeric-ridge) developed from the digital elevation model-derived metrics of aspect, slope, topographic position (the difference between the elevation value of an individual pixel and the average elevation in a 500- and 1,500-m-radius window around it), and relief. Landcover was classified to seven forest types derived from the NLCD: low-density residential, transitional-shrubland, deciduous, evergreen,

**Table 2.—continued**

| Data source | Species code[a] | | | | | |
|---|---|---|---|---|---|---|
| | BGGN | BWWA | BRTH | BHNU | CACH | CERW |
| DEM, NLCD, and FIA | | | | | | |
| Landform, landcover, and successional age class | 1 | 1 | 1 | 1 | 1 | 1 |
| NLCD and FIA | | | | | | |
| Early successional patch size (ha) | | 2 | | | | |
| NLCD and NHD | | | | | | |
| Occurrence of water | | | | | | |
| Distance (m) to water | | | | | | |
| NLCD | | | | | | |
| Forest patch size (ha) | 2 | | | | | 2 |
| Landscape composition (percent forest in 1-km radius) | | | | | | 3 |
| Landscape composition (percent forest in 10-km radius) | 3 | | 4 | | | |
| Occurrence of edge | 4 | | 2 | | | |
| Distance (m) to edge | | | | | | |
| Interspersion – 1 landcover class | | | | | | |
| Interspersion – 2 landcover classes | | | | | | |
| Connectivity (km) | | | | | | |
| Grass-open landcover | | | | | | |
| FIA | | | | | | |
| Basal area ($m^2$/ha) | 5 | | | | | |
| Hardwood basal area ($m^2$/ha) | | | | 4 | | |
| Pine basal area ($m^2$/ha) | | | | | | |
| Sawtimber (> 28 cm d.b.h.) tree density (trees/ha) | | | | | | |
| Large (> 50 cm d.b.h) tree density (trees/ha) | | | | | | |
| Large (> 35 cm d.b.h) pine density (trees/ha) | | | | | | |
| Dominant (> 76.2 cm d.b.h.) tree density (trees/ha) | | | | | | 4 |
| Midstory (11–25 cm d.b.h.) density (trees/ha) | | | | | | |
| Snag density (snags/ha) | | | | 2 | 2 | |
| Large (> 30 cm d.b.h.) snag density (snags/ha) | | | | | | |
| Canopy cover (percent) | | 3 | | | | 5 |
| Small stem (< 2.5 cm d.b.h.) density (stems/ha) | | | 3 | 3 | | |
| DEM | | | | | | |
| Slope | | | | | | |
| NHD | | | | | | |
| Distance (m) to stream | | | | | | |
| STATSGO | | | | | | |
| Soil texture | | | | | | |
| Soil moisture | | | | | | |

**continued**

mixed, orchard-vineyard, and woody wetlands. Finally, successional age class was delineated into five classes based on the average diameter at breast height (d.b.h.) of dominant trees in each stand, ultimately derived from FIA data: grass-forb (trees < 2.5 cm d.b.h.), shrub-seedling (2.5 to 7.5 cm), sapling (7.5 to 12.5 cm), pole (12.5 to 37.5 cm), and sawtimber (> 37.5 cm).

We assigned to each of the 105 unique landform, landcover, and successional age class combinations (three landform classes × seven forest type classes × five successional age classes) an SI value based on the relative habitat suitability rankings reported in the bird habitat matrices in Hamel (1992). These matrices qualitatively assess habitat suitability (marginal, suitable, optimal) for each bird species based on seral stage (4 classes) and forest type (23 classes). To adapt these matrices to our purposes, we crosswalked these forest types to our

**Table 2.—continued**

| Data source | Species code[a] | | | | | |
|---|---|---|---|---|---|---|
| | CHSW | CWWI | EAWP | FISP | GCFL | HOWA |
| DEM, NLCD, and FIA | | | | | | |
| Landform, landcover, and successional age class | | 1 | 1 | 1 | 1 | 1 |
| NLCD and FIA | | | | | | |
| Early successional patch size (ha) | | | | | | |
| NLCD and NHD | | | | | | |
| Occurrence of water | | | | | | |
| Distance (m) to water | | | | | | |
| NLCD | | | | | | |
| Forest patch size (ha) | | | | | | 4 |
| Landscape composition (percent forest in 1-km radius) | | | 2 | | | 5 |
| Landscape composition (percent forest in 10-km radius) | | | | | | |
| Occurrence of edge | | | | | | |
| Distance (m) to edge | | | | | 3 | |
| Interspersion – 1 landcover class | 1 | | | | | |
| Interspersion – 2 landcover classes | | 2 | | | | |
| Connectivity (km) | | | | | | |
| Grass-open landcover | | | | 4 | | |
| FIA | | | | | | |
| Basal area ($m^2$/ha) | | | | | | |
| Hardwood basal area ($m^2$/ha) | | | | | | |
| Pine basal area ($m^2$/ha) | | | | | | |
| Sawtimber (> 28 cm d.b.h.) tree density (trees/ha) | | | 3 | | | |
| Large (> 50 cm d.b.h) tree density (trees/ha) | | | | | | |
| Large (> 35 cm d.b.h) pine density (trees/ha) | | | | | | |
| Dominant (> 76.2 cm d.b.h.) tree density (trees/ha) | | | | | | |
| Midstory (11–25 cm d.b.h.) density (trees/ha) | | | | | | |
| Snag density (snags/ha) | | | | | 2 | |
| Large (> 30 cm d.b.h.) snag density (snags/ha) | | | | | | |
| Canopy cover (percent) | | | | 2 | | 3 |
| Small stem (< 2.5 cm d.b.h.) density (stems/ha) | | | | 3 | | 2 |
| DEM | | | | | | |
| Slope | | | | | | |
| NHD | | | | | | |
| Distance (m) to stream | | | | | | |
| STATSGO | | | | | | |
| Soil texture | | | | | | |
| Soil moisture | | | | | | |

**continued**

landform-landcover classes and adapted the four seral stages to our five successional age classes (Table 3). First, we identified which of the 23 forest types occurred in the CH or WGCP (seven types: Sandhills longleaf pine, oak-gum-cypress, elm-ash-cottonwood, loblolly pine-shortleaf pine, mixed pine-hardwood, oak-hickory, and cove hardwoods). We then assigned these forest types to specific landform and landcover combinations based on the physiography associated with these forest communities.

However, not all NLCD landcovers have an analogous forest types in the Hamel classification. For example, orchards-vineyards, low-density residential, and transitional-shrubland landcover types provide habitat for many priority species but do not have a specific forest type association. Therefore, we assigned to orchards-vineyards and low-density residential sites the same SI scores

**Table 2.—continued**

| | Species code[a] | | | | | |
|---|---|---|---|---|---|---|
| Data source | KEWA | LOWA | MIKI | NOBO | NOPA | OROR |
| DEM, NLCD, and FIA | | | | | | |
| Landform, landcover, and successional age class | 1 | 1 | 1 | 1 | 1 | 1 |
| NLCD and FIA | | | | | | |
| Early successional patch size (ha) | | | | | | |
| NLCD and NHD | | | | | | |
| Occurrence of water | | | | | | |
| Distance (m) to water | | | | | | |
| NLCD | | | | | | |
| Forest patch size (ha) | 3 | 5 | 2 | | 2 | |
| Landscape composition (percent forest in 1-km radius) | | 6 | | | 3 | 2 |
| Landscape composition (percent forest in 10-km radius) | 4 | | | | | |
| Occurrence of edge | | | | | | |
| Distance (m) to edge | | | | | | |
| Interspersion – 1 landcover class | | | | | | |
| Interspersion – 2 landcover classes | | | 3 | 5 | | |
| Connectivity (km) | | | | | | |
| Grass-open landcover | | | | 4 | | |
| FIA | | | | | | |
| Basal area ($m^2$/ha) | | | | | | 3 |
| Hardwood basal area ($m^2$/ha) | | | | 2 | | |
| Pine basal area ($m^2$/ha) | | | | 3 | | |
| Sawtimber (> 28 cm d.b.h.) tree density (trees/ha) | | | | | | |
| Large (> 50 cm d.b.h) tree density (trees/ha) | | | | | | |
| Large (> 35 cm d.b.h) pine density (trees/ha) | | | | | | |
| Dominant (> 76.2 cm d.b.h.) tree density (trees/ha) | | | 4 | | | |
| Midstory (11–25 cm d.b.h.) density (trees/ha) | | | | | | |
| Snag density (snags/ha) | | | | | | |
| Large (> 30 cm d.b.h.) snag density (snags/ha) | | | | | | |
| Canopy cover (percent) | | 3 | | | 4 | |
| Small stem (< 2.5 cm d.b.h.) density (stems/ha) | 2 | 4 | | | | |
| DEM | | | | | | |
| Slope | | | | | | |
| NHD | | | | | | |
| Distance (m) to stream | | 2 | | | | |
| STATSGO | | | | | | |
| Soil texture | | | | | | |
| Soil moisture | | | | | | |

**continued**

as those for deciduous landcovers on the assumption that orchards are composed primarily of deciduous species and low-density residential sites typically are planted with deciduous shade trees. Similarly, we assumed that transitional-shrubland sites are regenerating forests. Where there were transitional-shrubland pixels in floodplain-valley landforms, we assumed that they were hardwood forest regeneration. Thus, we assigned to them the same SI scores associated with deciduous habitats. On the higher and drier landforms, transitional-shrubland sites likely are dominated by oak and redcedar in the CH and pine in the WGCP, so we assigned to these sites the same SI scores as those for mixed and evergreen forest in each BCR, respectively (Table 3).

To assign SI scores to specific age classes, we used the relative habitat quality values reported in Hamel (1992) for grass-forb, shrub-seedling, and sawtimber seral stages. However, Hamel

**Table 2.—continued**

| | Species code[a] | | | | | |
|---|---|---|---|---|---|---|
| Data source | PABU | PIWO | PRAW | PROW | RCWO | RHWO |
| DEM, NLCD, and FIA | | | | | | |
| Landform, landcover, and successional age class | 1 | 1 | 1 | 1 | 1 | 1 |
| NLCD and FIA | | | | | | |
| Early successional patch size (ha) | | | 3 | | | |
| NLCD and NHD | | | | | | |
| Occurrence of water | | | | 2 | | |
| Distance (m) to water | | | | | | |
| NLCD | | | | | | |
| Forest patch size (ha) | | 3 | | 3 | 2 | |
| Landscape composition (percent forest in 1-km radius) | | 4 | | 4 | | |
| Landscape composition (percent forest in 10-km radius) | | | | | | |
| Occurrence of edge | | | 2 | | | 5 |
| Distance (m) to edge | 2 | | | | | |
| Interspersion – 1 landcover class | | | | | | |
| Interspersion – 2 landcover classes | 3 | | | | | |
| Connectivity (km) | | | | | 5 | |
| Grass-open landcover | | | | | | |
| FIA | | | | | | |
| Basal area ($m^2$/ha) | | | | | | |
| Hardwood basal area ($m^2$/ha) | | | | | 4 | |
| Pine basal area ($m^2$/ha) | | | | | 3 | |
| Sawtimber (> 28 cm d.b.h.) tree density (trees/ha) | | | | | | 4 |
| Large (> 50 cm d.b.h) tree density (trees/ha) | | | | | | |
| Large (> 35 cm d.b.h) pine density (trees/ha) | | | | | 6 | |
| Dominant (> 76.2 cm d.b.h.) tree density (trees/ha) | | | | | | |
| Midstory (11–25 cm d.b.h.) density (trees/ha) | | | | | | |
| Snag density (snags/ha) | | | | 5 | | 2 |
| Large (> 30 cm d.b.h.) snag density (snags/ha) | | 2 | | | | 3 |
| Canopy cover (percent) | | | 5 | | | |
| Small stem (< 2.5 cm d.b.h.) density (stems/ha) | 4 | | 4 | | | |
| DEM | | | | | | |
| Slope | | | | | | |
| NHD | | | | | | |
| Distance (m) to stream | | | | | | |
| STATSGO | | | | | | |
| Soil texture | | | | | | |
| Soil moisture | | | | | | |

**continued**

combined sapling- and pole-size trees into a single class, whereas we separated these two successional age classes (a segregation we believed was more appropriate for many of our species). To tease apart the SI scores for sapling and pole age classes, we averaged the value for sapling-pole with shrub-seedling (for sapling) or sawtimber (for pole). This approach assumes that sapling and pole stands have an equal weighting by Hamel in assessing the relative habitat quality for the aggregate age class, and that there is a linear relationship across age classes that allows us to discern the relative influence of each by simple averaging.

After crosswalking Hamel's forest types and seral stages to our landform-landcover-successional age class matrix, we assigned SI scores to each unique combination based on Hamel's qualitative assessments. Combinations considered optimal (Hamel 1992) were assigned a value of 1.000;

**Table 2.—continued**

| | Species code[a] | | | | | |
|---|---|---|---|---|---|---|
| Data source | SWWA | STKI | WPWI | WEVI | WOTH | WEWA |
| DEM, NLCD, and FIA | | | | | | |
| Landform, landcover, and successional age class | 1 | 1 | 1 | 1 | 1 | 1 |
| NLCD and FIA | | | | | | |
| Early successional patch size (ha) | | | | | | |
| NLCD and NHD | | | | | | |
| Occurrence of water | | | | | | |
| Distance (m) to water | | | | | | |
| NLCD | | | | | | |
| Forest patch size (ha) | 2 | 2 | | | 2 | 3 |
| Landscape composition (percent forest in 1-km radius) | 3 | | | | 3 | 4 |
| Landscape composition (percent forest in 10-km radius) | | | | | | |
| Occurrence of edge | | | | 2 | | |
| Distance (m) to edge | | | | | | |
| Interspersion – 1 landcover class | | 3 | | | | |
| Interspersion – 2 landcover classes | | | 2 | | | |
| Connectivity (km) | | | | | | |
| Grass-open landcover | | | | | | |
| FIA | | | | | | |
| Basal area ($m^2$/ha) | | | | | | |
| Hardwood basal area ($m^2$/ha) | | | | | | |
| Pine basal area ($m^2$/ha) | | | | | | |
| Sawtimber (> 28 cm d.b.h.) tree density (trees/ha) | | | | | | |
| Large (> 50 cm d.b.h) tree density (trees/ha) | | | | | | |
| Large (> 35 cm d.b.h) pine density (trees/ha) | | | | | | |
| Dominant (> 76.2 cm d.b.h.) tree density (trees/ha) | | 4 | | | | |
| Midstory (11–25 cm d.b.h.) density (trees/ha) | | | | | | |
| Snag density (snags/ha) | | | | | | |
| Large (> 30 cm d.b.h.) snag density (snags/ha) | | | | | | |
| Canopy cover (percent) | | | | 3 | 5 | |
| Small stem (< 2.5 cm d.b.h.) density (stems/ha) | 4 | | | 4 | 4 | 5 |
| DEM | | | | | | |
| Slope | | | | | | 2 |
| NHD | | | | | | |
| Distance (m) to stream | | | | | | |
| STATSGO | | | | | | |
| Soil texture | | | | | | |
| Soil moisture | | | | | | |

**continued**

those considered suitable were assizned a value of 0.667; and those considered marginal had a value of 0333. We assumed that forest types and age classes not assigned a qualitative habitat ranking were not used and assigned to these combinations an SI score of zero. Where a landform-landcover type was represented by more than one of Hamel's forest types, SI values for the forest types were averaged. For example, deciduous landcover on floodplain-valley landforms are associated with cove hardwood and elm-ash-cottonwood forest communities. Cove hardwood is suitable (SI = 0.667) for the Acadian flycatcher but elm-ash-cottonwood is optimal (SI = 1.000). Thus, this landform-landcover type combination is assigned a base SI score of 0.834 (i.e., 1.667/2) prior to adjusting for successional age class (Table 4). Finally, we standardized all SI scores in the matrix to ensure that the maximum value was 1.000.

**Table 2.—continued**

| | Species code[a] | | | |
|---|---|---|---|---|
| Data source | YBCU | YBCH | YTVI | YTWA |
| DEM, NLCD, and FIA | | | | |
| Landform, landcover, and successional age class | 1 | 1 | 1 | 1 |
| NLCD and FIA | | | | |
| Early successional patch size (ha) | | 3 | | |
| NLCD and NHD | | | | |
| Occurrence of water | | | | |
| Distance (m) to water | | | | 3 |
| NLCD | | | | |
| Forest patch size (ha) | 5 | | 2 | |
| Landscape composition (percent forest in 1-km radius) | | | 3 | 4 |
| Landscape composition (percent forest in 10-km radius) | 4 | | | |
| Occurrence of edge | 2 | 2 | | |
| Distance (m) to edge | | | | |
| Interspersion – 1 landcover class | | | | |
| Interspersion – 2 landcover classes | | | | |
| Connectivity (km) | | | | |
| Grass-open landcover | | | | |
| FIA | | | | |
| Basal area ($m^2$/ha) | | | | |
| Hardwood basal area ($m^2$/ha) | | | | |
| Pine basal area ($m^2$/ha) | | | | |
| Sawtimber (> 28 cm d.b.h.) tree density (trees/ha) | | | | |
| Large (> 50 cm d.b.h) tree density (trees/ha) | | | | 2 |
| Large (> 35 cm d.b.h) pine density (trees/ha) | | | | |
| Dominant (> 76.2 cm d.b.h.) tree density (trees/ha) | | | | |
| Midstory (11–25 cm d.b.h.) density (trees/ha) | 3 | | | |
| Snag density (snags/ha) | | | | |
| Large (> 30 cm d.b.h.) snag density (snags/ha) | | | | |
| Canopy cover (percent) | | | 4 | |
| Small stem (< 2.5 cm d.b.h.) density (stems/ha) | | 4 | | |
| DEM | | | | |
| Slope | | | | |
| NHD | | | | |
| Distance (m) to stream | | | | |
| STATSGO | | | | |
| Soil texture | | | | |
| Soil moisture | | | | |

[a] Pyle and DeSante 2003; see Table 1.

Similarly, we directly assigned SI scores to individual classes for other discrete environmental variables (e.g., occurrence of water). For continuous environmental variables (e.g., canopy cover), we used CurveExpert 1.38 software (Hyams 2001)[1] to fit smoothed functions through known data points derived from the literature that quantify the relationship between each specific environmental factor and HSI scores for particular species. Information sources, assumptions, and functions (type and equation) are detailed in the model accounts.

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[1] The use of trade, firm, or corporation names in this publication is for the information and convenience of the reader. Such use does not constitute an official endorsement or approval by the U.S. Department of Agriculture or Forest Service of any product or service to the exclusion of others that may be suitable.

**Table 3.—Crosswalk between landform-landcover class combinations and vegetation types defined in Hamel (1992)**

| Landform | Landcover type | Hamel vegetation type[a] |
|---|---|---|
| Floodplain-valley | Low-density residential | Same as deciduous |
| | Transitional-shrubland | Same as deciduous |
| | Deciduous | Cove hardwoods<br>Elm-ash-cottonwood |
| | Evergreen | Loblolly pine-shortleaf pine |
| | Mixed | Mixed pine-hardwood |
| | Orchards-vineyards | Same as deciduous |
| | Woody wetlands | Oak-gum-cypress<br>Elm-ash-cottonwood |
| Terrace-mesic | Low-density residential | Same as deciduous |
| | Transitional-shrubland | Same as mixed in Central Hardwoods, same as evergreen in West Gulf Coastal Plain/Ouachitas |
| | Deciduous | Oak-hickory<br>Cove hardwoods |
| | Evergreen | Loblolly pine-shortleaf pine |
| | Mixed | Mixed pine-hardwood |
| | Orchards-vineyards | Same as deciduous |
| | Woody wetlands | Elm-ash-cottonwood |
| Xeric-ridge | Low-density residential | Same as deciduous |
| | Transitional-shrubland | Same as Mixed in Central Hardwoods, same as evergreen in West Gulf Coastal Plain/Ouachitas |
| | Deciduous | Oak-hickory |
| | Evergreen | Loblolly pine-shortleaf pine. Also includes Sandhills longleaf pine in West Gulf Coastal Plain/Ouachitas |
| | Mixed | Mixed pine-hardwood |
| | Orchards-vineyards | Same as deciduous |
| | Woody wetlands | Elm-ash-cottonwood |

[a] Hamel (1992).

To calculate the overall HSI score, we determined the geometric mean of SI scores for site-scale and landscape-scale variables separately and then the geometric mean of these means together. Use of the geometric mean follows recommendations from the published standards for development of HSI models (USDI Fish and Wildl. Serv. 1981). The equal weighting of individual functions within a spatial scale assumes that all variables are required for a habitat to be suitable and that all variables are nonsubstitutable. Further, the equal weighting of functions across scales assumes that site and landscape variables are equally important. The notable exception to use of the geometric mean was for species where both forest patch size and percent forest in the landscape are included as model parameters. In these cases, we used the maximum SI score from these two variables to account for the use of small forest patches by area-sensitive species when small patches are embedded in predominantly forested landscapes (Rosenberg and others 1999). For each species, we solicited at least five reviewers with an intimate knowledge of the habitat requirements of at least one species. Each reviewer received a standard questionnaire requesting feedback on the appropriateness of the functions included in the model. We revised models based on reviewers' comments.

## Model Testing

To test the HSI models for reliability, we followed the three-stage framework (calibration, verification, and validation) outlined by Brooks (1997). We first ensured that the equations

**Table 4.—Initial assignment of suitability index scores for Acadian flycatcher habitat to landform, landcover type, and successional age classes based on Hamel (1992)**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.834 | 0.834 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.333 | 0.333 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 1.000 | 1.000 | 1.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.667 | 0.667 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.333 | 0.333 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.834 | 0.834 | 1.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.333 | 0.333 | 0.667 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.333 | 0.333 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.834 | 0.834 | 1.000 |

used to predict SI scores resulted in the full potential range of SI scores given the habitat conditions within each BCR (i.e., calibration). We then used Spearman rank correlation to compare HSI scores to abundance estimates from Breeding Bird Survey (BBS) data summarized by ecological subsection (i.e., verification). We ranked subsections by HSI score and BBS abundance for each species and within each BCR independently to compensate for geographical differences in these regions not explicitly incorporated in the HSI models. We assessed correlations between these variables based on all subsections and based solely on subsections within which each species was detected. The former analysis provides insight into the overall model performance; the latter addresses the potential bias associated with correctly predicting the absence of a rare species in many subsections.

Following verification, we validated HSI models by developing species-specific generalized linear models that predicted abundance (as indexed by BBS data) from HSI and BCR predictor variables. We considered HSI models validated if the general linear model was significant ($P < 0.100$) and the coefficient on the HSI predictor variable was both significant ($P < 0.100$) and positive. Detailed results of these analyses are documented in Tirpak and others (2009a).

# MODEL ACCOUNTS

## Acadian Flycatcher

John J. Mosesso, images.nbii.gov

### Status

The Acadian flycatcher (*Empidonax virescens*) is a long-distance migrant found throughout most of the eastern United States. While populations have declined in the northern portion of its range (particularly the Appalachians) over the last 40 years, populations in the South, particularly along the Atlantic and East Gulf Coastal Plains, have increased (Sauer and others 2005). However, the Acadian flycatcher has declined in the WGCP (Table 5), and the FWS classifies this species as a Bird of Conservation Concern in the WGCP (Table 1). Similarly, PIF considers the Acadian flycatcher as a planning and responsibility species in the CH (regional combined score of 16). In the WGCP, the flycatcher has a regional combined score of 17, warranting management attention (Table 1).

### Natural History

The Acadian flycatcher is a forest-interior species associated with water throughout most of its range: bottomland hardwood and cypress forests in the Southeast and riparian forests and ravines in the deciduous forests of the Midwest and Northeast (Whitehead and Taylor 2002). This species is found in numerous forest types and uses a variety of tree species for nesting. However, this bird typically is associated with mesic forest stands and avoids upland oak-hickory sites (Klaus and others 2005). Breeding territories are small and average 1 ha (Woolfenden and others 2005). The Acadian flycatcher typically nests in midstory trees and large shrubs in mature forests. Canopy cover typically is dense (> 95 percent; Wilson and Cooper 1998), and the understory usually is sparse (Bell and Whitmore 2000, Wood and others 2004).

The Acadian flycatcher is particularly susceptible to forest fragmentation. Aquilani and Brewer (2004) found this species only in forest tracts larger than 55 ha in north-central Mississippi. Blake and Karr (1987) did not observe the Acadian flycatcher in woodlots smaller than 24 ha. In east Texas, the Acadian flycatcher was absent from riparian buffer strips less than 70 m wide (Conner and others 2004). Results were similar in Missouri (Peak and others 2004) and Indiana (Ford and others 2001).

Even in large forested tracts (> 600 ha), nest predation and parasitism rates may be 10 to 20 percent higher if the surrounding landscape is highly fragmented. Nevertheless, Fauth and Cabe (2005) did not observe significant effects of parasitism on a Blue Ridge study site where 75 percent of the landscape was forested, including 45 percent more than 250 m from an edge. Disturbance, whether natural (e.g., tornado or pest outbreak) or anthropogenic (e.g., silvicultural treatments—thinning, selective harvesting, clearcutting, and prescribed burning) reduced the abundance and productivity of the Acadian flycatcher in most landscapes (Artman and others 2001, Duguay and others 2001, Robinson and Robinson 2001, Twedt and others 2001, Prather and Smith 2003, Blake 2005).

**Table 5.—Trend estimates (percent change per year) for 40 priority landbird species in the Central Hardwoods and West Gulf Coastal Plain/Ouachitas Bird Conservation Regions, 1967 to 2004 (Sauer and others 2005)**

| | Central Hardwoods | | | West Gulf Coastal Plain/Ouachitas | | |
|---|---|---|---|---|---|---|
| Species | Trend | *P* | $n^a$ | Trend | *P* | *n* |
| Acadian flycatcher | -0.3 | 0.56 | 107 | -2.0 | 0.05 | 67 |
| American woodcock | -9.1 | 0.35 | 3 | --[b] | -- | -- |
| Bachman's sparrow | -- | -- | -- | -7.8 | 0.00 | 27 |
| Bell's vireo | -3.2 | 0.49 | 18 | -4.7 | 0.03 | 14 |
| Bewick's wren | -6.5 | 0.00 | 61 | 0.8 | 0.88 | 11 |
| Black-and-white warbler | 2.3 | 0.21 | 50 | -2.9 | 0.01 | 60 |
| Blue-gray gnatcatcher | -1.0 | 0.26 | 118 | -0.9 | 0.36 | 75 |
| Blue-winged warbler | -4.0 | 0.01 | 62 | -- | -- | -- |
| Brown thrasher | -1.4 | 0.00 | 125 | -1.4 | 0.01 | 64 |
| Brown-headed nuthatch | -- | -- | -- | -1.4 | 0.18 | 52 |
| Carolina chickadee | 0.2 | 0.70 | 123 | -2.0 | 0.00 | 77 |
| Cerulean warbler | -6.3 | 0.00 | 34 | -9.5 | 0.00 | 5 |
| Chimney swift | -2.6 | 0.00 | 124 | -1.1 | 0.15 | 76 |
| Chuck-will's-widow | -0.9 | 0.19 | 64 | -1.3 | 0.04 | 60 |
| Eastern wood-pewee | -1.4 | 0.00 | 124 | -4.9 | 0.00 | 75 |
| Field sparrow | -3.2 | 0.00 | 125 | -3.7 | 0.01 | 45 |
| Great crested flycatcher | -0.8 | 0.09 | 123 | -1.3 | 0.04 | 77 |
| Hooded warbler | 2.7 | 0.08 | 31 | -3.1 | 0.35 | 60 |
| Kentucky warbler | -0.4 | 0.32 | 108 | -2.2 | 0.00 | 73 |
| Louisiana waterthrush | 2.6 | 0.02 | 66 | -1.3 | 0.49 | 28 |
| Mississippi kite | 16.3 | 0.16 | 2 | 6.4 | 0.21 | 16 |
| Northern bobwhite | -3.1 | 0.00 | 125 | -4.4 | 0.00 | 75 |
| Northern parula | 3.7 | 0.00 | 95 | -2.5 | 0.17 | 53 |
| Orchard oriole | -0.9 | 0.01 | 124 | -3.0 | 0.01 | 75 |
| Painted bunting | 19.8 | 0.61 | 5 | -0.6 | 0.48 | 63 |
| Pileated woodpecker | 1.8 | 0.01 | 112 | -0.9 | 0.14 | 72 |
| Prairie warbler | -2.6 | 0.00 | 94 | -4.4 | 0.00 | 60 |
| Prothonotary warbler | 0.0 | 0.98 | 52 | -5.8 | 0.00 | 53 |
| Red-cockaded woodpecker | -- | -- | -- | 9.0 | 0.00 | 6 |
| Red-headed woodpecker | -1.0 | 0.09 | 115 | -3.2 | 0.00 | 68 |
| Swainson's warbler | -- | -- | -- | 23.5 | 0.23 | 26 |
| Swallow-tailed kite | -- | -- | -- | -- | -- | -- |
| Whip-poor-will | -1.8 | 0.05 | 71 | 6.6 | 0.22 | 11 |
| White-eyed vireo | -0.4 | 0.20 | 120 | -0.8 | 0.19 | 76 |
| Wood thrush | -0.7 | 0.05 | 118 | -1.4 | 0.05 | 67 |
| Worm-eating warbler | 0.4 | 0.77 | 44 | -2.3 | 0.51 | 28 |
| Yellow-billed cuckoo | -1.9 | 0.00 | 125 | -1.1 | 0.00 | 77 |
| Yellow-breasted chat | -1.9 | 0.00 | 125 | 1.3 | 0.01 | 75 |
| Yellow-throated vireo | 0.9 | 0.25 | 99 | 1.1 | 0.38 | 62 |
| Yellow-throated warbler | 3.8 | 0.00 | 76 | -0.9 | 0.65 | 43 |

[a] Number of Breeding Bird Survey routes on which trend estimate is based.
[b] No trend estimate available.

**Table 6.—Relationship of landform, landcover type, and successional age class to suitability index scores for Acadian flycatcher habitat**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.050 | 0.917 | 1.000 |
| | Deciduous | 0.000 | 0.000 | 0.050 | 0.917 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.017 | 0.167 | 0.333 |
| | Mixed | 0.000 | 0.000 | 0.017 | 0.333 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.050 | 1.000 | 1.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.017 | 0.333 | 0.333 |
| | Deciduous | 0.000 | 0.000 | 0.042 | 0.667 | 0.834 |
| | Evergreen | 0.000 | 0.000 | 0.017 | 0.167 | 0.333 |
| | Mixed | 0.000 | 0.000 | 0.017 | 0.333 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.050 | 1.000 | 1.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.017 | 0.333 | 0.333 |
| | Deciduous | 0.000 | 0.000 | 0.033 | 0.500 | 0.667 |
| | Evergreen | 0.000 | 0.000 | 0.017 | 0.167 | 0.333 |
| | Mixed | 0.000 | 0.000 | 0.017 | 0.333 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.050 | 1.000 | 1.000 |

## Model Description

Our Acadian flycatcher model includes seven variables related to density: landform, landcover type, successional age class, distance to water, canopy cover, forest patch size, and percent forest in a 1-km radius window.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 6). We directly assigned SI scores to these combinations on the basis of habitat suitability data from Hamel (1992) on the relative quality of different vegetation types and successional stages for the Acadian flycatcher. However, we reduced SI scores for sapling and evergreen habitats on the basis of data from Hazler (1999).

Because the Acadian flycatcher typically is found near water (Whitehead and Taylor 2002), we fit an inverse logistic function to describe the relationship between SI scores for this species and increasing distance to water (SI2; Fig 2). The flycatcher often aligns at least one edge of its 1-ha territory along a stream or wetland (Woolfenden and others 2005).

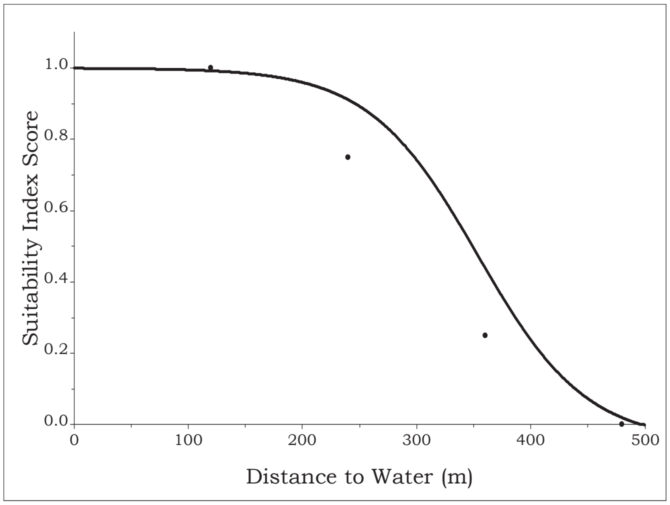


Figure 2.—Relationship between distance to water and suitability index (SI) scores for Acadian flycatcher habitat. Equation: SI score = 1 - (1.049 / (1 + (1664.953 * e $^{-0.021 * \text{distance to water}}$))).

**Table 7.—Relationship between distance to water and suitability index (SI) scores for Acadian flycatcher habitat**

| Distance to water (m)[a] | SI score |
|---|---|
| 0[b] | 1.00 |
| 120[c] | 1.00 |
| 240[b] | 0.75 |
| 360[b] | 0.25 |
| 480[b] | 0.00 |

[a] Water defined as streams from the National Hydrography Dataset (medium resolution) or classified as water, woody wetlands, or emergent herbaceous wetlands in the National Land Cover Dataset.

[b] Assumed value.

[c] Woolfenden and others (2005).

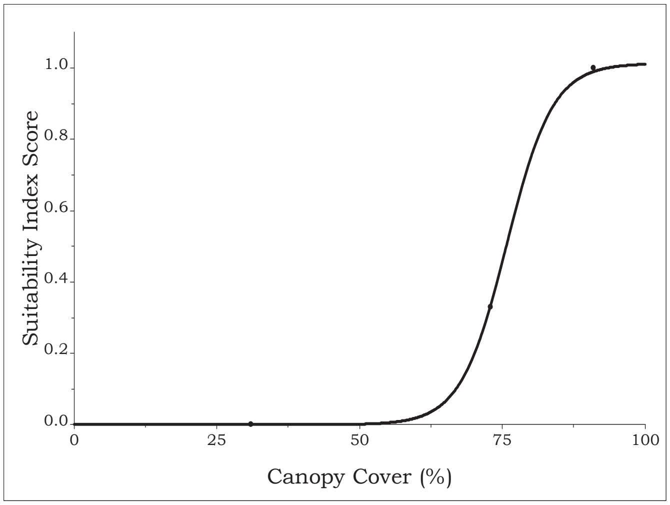


Figure 3.—Relationship between canopy cover and suitability index (SI) scores for Acadian flycatcher habitat. Equation: SI score = 1.013 / (1.000 + (144082770 * e $^{-0.248 * \text{canopy cover}}$)).

**Table 8.—Relationship between canopy cover and suitability index (SI) scores for Acadian flycatcher habitat**

| Canopy cover (percent) | SI score |
|---|---|
| 0[a] | 0.00 |
| 31[b] | 0.00 |
| 73[b] | 0.33 |
| 91[b] | 1.00 |
| 100[a] | 1.00 |

[a] Assumed value.

[b] Prather and Smith (2003).

Assuming a circular home range, the diameter of the home range (112.8 m) represents the farthest distance from water a bird could be within the home range. On the basis of this assumption, we assigned all locations less than 120 m from water SI scores of 1.000 (Table 7). The Acadian flycatcher also uses sites that are more than 120 m from water but generally are found at lower densities there. Thus, we considered areas 360 m from water (a distance of three home range diameters) as having an SI score that is one-quarter of the optimal value (0.250) and sites at least 480 m from water as nonhabitat (SI score of zero).

The habitat suitability model for the Acadian flycatcher also included canopy closure (SI3) as a variable because of the strong affinity of this species for closed-canopy forests (Prather and Smith 2003). For this variable, we used a logistic function (Fig. 3) to extrapolate between known break points in the canopy cover-relative density relationship (Table 8).

We also included forest patch size (SI4) as a variable because of the sensitivity of the Acadian flycatcher to fragmentation (Robbins and others 1989) and increasing edge density (Parker and others 2005). We used a logarithmic function (Fig. 4) to describe the relatively quick increase in suitability of a forest patch with increasing area (Robbins and others 1989) (Table 9). We assumed that 312 ha, the minimum forest patch size on which Wallendorf and others (2007) always observed the Acadian flycatcher, was representative of optimal habitat (SI score = 1.000). Nevertheless, the effects of forest patch size on suitability are influenced by the percentage of forest in the landscape. In predominantly forested landscapes, small forest patches that may not be used in predominantly nonforested landscapes may provide habitat due to their proximity to large forest blocks (Rosenberg and others 1999). To capture this relationship, we fit a logistic function (Fig. 5) to data (Table 10) derived from Donovan and others (1997), who observed differences in predator and brood parasite communities among highly fragmented (< 15 percent), moderately fragmented (45 to 50 percent), and lightly fragmented (> 90 percent forest) landscapes. We assumed that the midpoints between these classes (30 and 70 percent forest) defined the specific cutoffs for poor (SI score ≤ 0.10) and excellent (SI score ≥ 0.90) habitat, respectively. We used the maximum value of SI4 or SI5 to assess area sensitivity and to account for small patches in predominantly forested landscapes and large patches in predominantly non-forested landscapes.

To calculate the overall HSI score, we determined the geometric mean of SI scores for forest structure attributes (SI1 and SI3) and landscape attributes (maximum value of SI4 or SI5 and SI2) separately and then the geometric mean of these means together.

$$\text{Overall HSI} = ((\text{SI1} * \text{SI3})^{0.500} * (\text{Max}(\text{SI4 or SI5}) * \text{SI2})^{0.500})^{0.500}$$

## Verification and Validation

The Acadian flycatcher was found in all 88 subsections of the CH and WGCP. Spearman rank correlation identified a significant ($P \leq 0.001$) positive relationship ($r_s = 0.47$) between average HSI score and mean BBS abundance across subsections. The generalized linear model predicting BBS abundance from BCR and HSI for the Acadian flycatcher was significant ($P = 0.095$; $R^2 = 0.054$), and the coefficient on the HSI predictor variable was both positive ($\beta = 4.250$) and significantly different from zero ($P = 0.043$). Therefore, we considered the HSI model for the Acadian flycatcher both verified and validated (Tirpak and others 2009a).

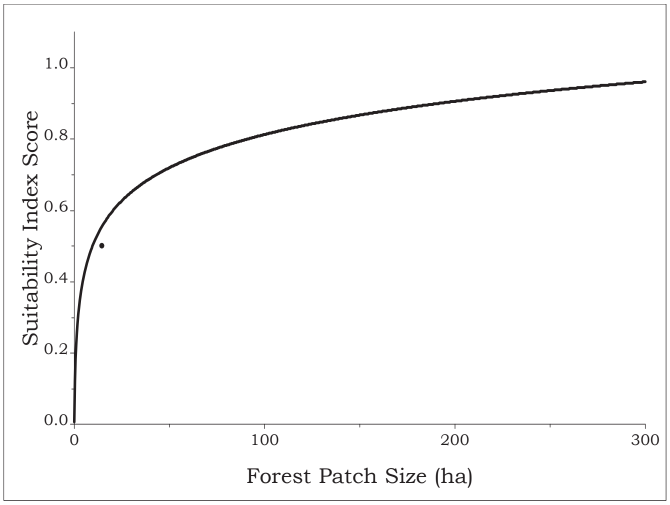


Figure 4.—Relationship between forest patch size and suitability index (SI) scores for Acadian flycatcher habitat. Equation: SI score = 0.174 * ln(forest patch size) + 0.010.

**Table 9.—Relationship between forest patch size and suitability index (SI) scores for Acadian flycatcher habitat**

| Forest patch size (ha) | SI score |
|---|---|
| 0.2[a] | 0.0 |
| 15[a] | 0.5 |
| 312[b] | 1.0 |

[a] Robbins and others (1989).

[b] Wallendorf and others (2007).

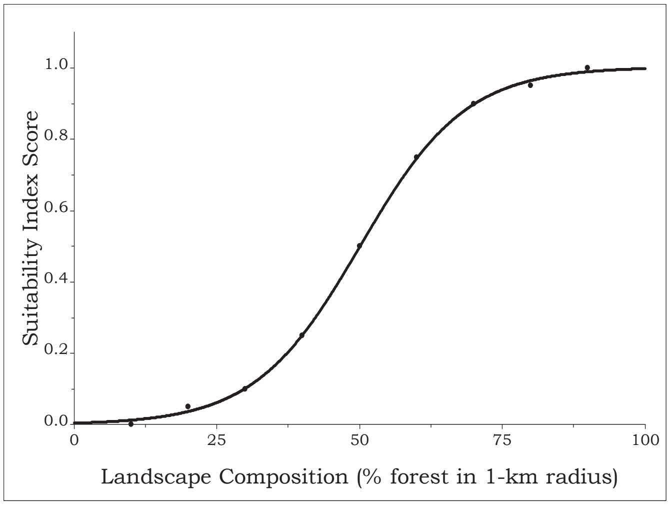


Figure 5.—Relationship between landscape composition and suitability index (SI) scores for Acadian flycatcher habitat. Equation: SI score = 1.005 / (1.000 + (221.816 * e $^{-0.108 * \text{(landscape composition)}}$)).

**Table 10.—Relationship between local landscape composition (percent forest in 1-km radius) and suitability index (SI) scores for Acadian flycatcher habitat**

| Local landscape composition | SI score |
|---|---|
| 0[a] | 0.00 |
| 10[a] | 0.00 |
| 20[a] | 0.05 |
| 30[b] | 0.10 |
| 40[a] | 0.25 |
| 50[b] | 0.50 |
| 60[a] | 0.75 |
| 70[b] | 0.90 |
| 80[a] | 0.95 |
| 90[b] | 1.00 |
| 100[a] | 1.00 |

[a] Assumed that value.

[b] Dononvan and others (1997).

## American Woodcock


U.S. Fish & Wildlife Service

### Status

The American woodcock (*Scolopax minor*) is a popular gamebird found throughout the eastern United States and southeastern Canada. Although this species breeds primarily in the northern portion of its continental range, small numbers breed regularly throughout the wintering range in the Southeast. Singing ground surveys and wing collections from northern latitudes in the Central United States document annual 1.8 percent declines in woodcock since 1968 (Kelley 2003). The status of the relatively small breeding population in the Southeast is unknown.

### Natural History

The American woodcock breeds in early successional habitat throughout its range (Keppie and Whiting 1994). Typically, these young forest stands are on moist, uncompacted soils that allow the woodcock to probe for earthworms, the bird's preferred food (Steketee 2000). Equally important is an interspersion of the forest with openings that provide sites for both courtship displays and roosting (Sepik and Derleth 1993). Openings used by woodcock in Maine generally were at least 1.2 ha (Dunford and Owen 1973). Given the affinity of the woodcock for openings and early successional habitat, Sprankle and others (2000) recommended even-age forest management in rotational blocks to ensure that both habitat requirements are met.

Most of the available quantitative information on breeding habitat for the American woodcock is from the Northeast, particularly Maine and Pennsylvania (Straw and others 1986, McAuley and others 1996). Shrub cover generally is high (75 to 87 percent; Morgenweck 1977), while overstory cover typically is moderate (50 to 64 percent; Dunford and Owen 1973, Gregg and others 2000). Nests are in young forest stands (Morgenweck 1977). McAuley and others (1996) compared nest sites to random sites and found lower basal area and fewer coniferous saplings, but higher densities of deciduous saplings and shrub stems around nests sites. Young broods inhabit young to mid-age forest interspersed with openings; older broods occupy sites with greater basal area but fewer mature trees (Morgenweck 1977).

Many habitat variables have been associated with the presence of woodcock (Storm and others 1995; Klute and others 2002). Landcover variables were the best predictors at fine scales whereas indices of landscape heterogeneity were the most important predictors at large spatial scales (Klute and others 2000). Murphy and Thompson (1993) developed a model to predict the density of males on singing grounds in central Missouri that contained small stem density (≤ 2.5 cm d.b.h.), tree density (> 2.5 cm d.b.h.), and field size as predictor variables.

**Table 11.—Relationship of landform, landcover type, and successional age class to suitability index scores for American woodcock habitat**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 1.000 | 0.667 | 0.333 |
| | Deciduous | 0.000 | 0.000 | 1.000 | 0.667 | 0.333 |
| | Evergreen | 0.000 | 0.000 | 0.500 | 0.250 | 0.125 |
| | Mixed | 0.000 | 0.000 | 0.667 | 0.333 | 0.167 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.667 | 0.333 | 0.167 |
| | Woody wetlands | 0.000 | 0.000 | 1.000 | 0.667 | 0.333 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.834 | 0.500 | 0.250 |
| | Deciduous | 0.000 | 0.000 | 0.834 | 0.500 | 0.250 |
| | Evergreen | 0.000 | 0.000 | 0.400 | 0.200 | 0.100 |
| | Mixed | 0.000 | 0.000 | 0.500 | 0.250 | 0.125 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.500 | 0.250 | 0.125 |
| | Woody wetlands | 0.000 | 0.000 | 0.834 | 0.500 | 0.250 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.750 | 0.400 | 0.167 |
| | Deciduous | 0.000 | 0.000 | 0.750 | 0.400 | 0.167 |
| | Evergreen | 0.000 | 0.000 | 0.333 | 0.167 | 0.083 |
| | Mixed | 0.000 | 0.000 | 0.400 | 0.200 | 0.100 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.500 | 0.417 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.750 | 0.400 | 0.167 |

## Model Description

The American woodcock HSI model includes seven variables: landform, landcover, successional age class, small stem density (< 2.5 cm d.b.h.), composition of appropriately sized foraging-nesting and courtship-roosting habitat patches in the landscape, soil moisture, and soil texture.

The first suitability function combines landform, landcover type, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 11). Because the woodcock prefers moist habitats with high deciduous stem densities, we assigned the highest SI scores to sapling-aged transitional, deciduous, and woody wetland cover types in floodplain-valley landforms. We considered mixed and evergreen forests as well as xeric-ridge landforms as poor habitat for the American woodcock.

We included small stem density (SI2) as a model function because the woodcock relies on vertical structure to provide security from predators as it forages, nests, and loafs during the day. McAuley and others (1996) summarized habitat attributes around woodcock nest sites from seven studies in which stem density ranged from 5,051 to 49,250 stems per ha. Due to the relatively small sample size and the lack of geographic representation within the samples

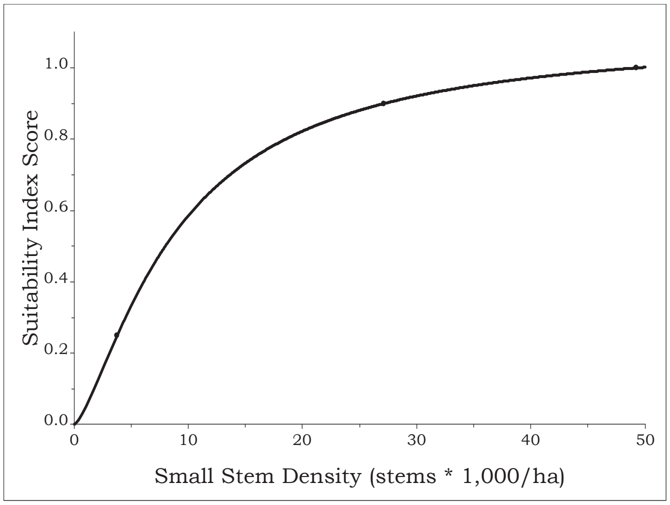


Figure 6.—Relationship between small stem (< 2.5 cm d.b.h.) density (stems*1000/ha) and suitability index (SI) scores for American woodcock habitat. Equation: SI score = 1.029 * $(0.998 - e^{-0.076 * (\text{small stem density} / 1000)})$.

**Table 12.—Influence of small stem (< 2.5 cm d.b.h.) density (stems*1,000/ha) on suitability index (SI) scores for American woodcock habitat**

| Small stem density | SI score |
|---|---|
| 0[accc] | 0.00 |
| 3.767[b] | 0.25 |
| 27.125[c] | 0.90 |
| 49.250[d] | 1.00 |

[a] Assumed value.
[b] Murphy and Thompson (1993).
[c] McAuley and others (1996).
[d] Coon and others (1982).

(both New York and Pennsylvania are represented twice), we used the midpoint of this range rather than the average to summarize these data. With three of the studies observing stem densities of at least 44,000 and three observing densities of approximately 14,000 stems per ha (+/- 600 stems/ha), we believed there was adequate evidence to assign to the midpoint of this range (27,125 stems/ha) a higher SI score than average (0.500). Therefore, we assigned 27,125 stems per ha an SI score of 0.900, the maximum stem density (49,250) an SI score of 1.000 and the minimum density (3,767 stems/ha, as reported by Murphy and Thompson [1993]) an SI score of 0.250 (Table 12). We fit a logistic function through these data points to quantify the small stem density-SI score relationship (Fig. 6).

The next two variables relate to the minimum size of habitat patches used by the American woodcock. Movement rates within diurnal foraging and nesting habitats often are low, resulting in small diurnal home ranges (≤ 0.3 ha; Hudgins and others 1985). Conversely, the woodcock displays and roosts in relatively large openings at night (≥ 1.6 ha; Keppie and Whiting 1994). We used these data to establish minimum area thresholds for forests and openings, respectively. Nevertheless, the ultimate suitability of either of these habitat types is related to their interspersion with one another, as the woodcock requires both. Ideally, these habitats should be separated by less than 400 m (Hudgins and others 1985) even though the average home range may be at least 74 ha (485-m radius; Keppie and Whiting 1994). Because home ranges may encompass areas of nonhabitat, the American woodcock sometimes is found where the proportion of these habitat types within a typical home range is relatively small (e.g., 0.1; Table 13). We assumed that the woodcock derives greater benefit from increasing proportions of early successional forest habitat than field habitat within its home ranges due to greater foraging opportunities and increased protection from predators. Thus, our table defining the relationship between landscape composition (SI3)

**Table 13.—Suitability index scores for American woodcock habitat based on composition of open and forest habitat within 500-m radius**

| | Proportion open[a] | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Proportion forest[b] | 0.00 | 0.10 | 0.20 | 0.30 | 0.40 | 0.50 | 0.60 | 0.70 | 0.80 | 0.90 | 1.00 |
| 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| 0.10 | 0.05 | 0.05 | 0.05 | 0.05 | 0.05 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | |
| 0.20 | 0.10 | 0.10 | 0.10 | 0.10 | 0.10 | 0.05 | 0.05 | 0.05 | 0.00 | | |
| 0.30 | 0.20 | 0.20 | 0.20 | 0.20 | 0.20 | 0.10 | 0.10 | 0.05 | | | |
| 0.40 | 0.40 | 0.40 | 0.40 | 0.40 | 0.40 | 0.20 | 0.10 | | | | |
| 0.50 | 0.60 | 0.60 | 0.60 | 0.60 | 0.60 | 0.40 | | | | | |
| 0.60 | 0.80 | 0.80 | 0.80 | 0.80 | 0.80 | | | | | | |
| 0.70 | 1.00 | 1.00 | 1.00 | 1.00 | | | | | | | |
| 0.80 | 1.00 | 1.00 | 1.00 | | | | | | | | |
| 0.90 | 1.00 | 1.00 | | | | | | | | | |
| 1.00 | 1.00 | | | | | | | | | | |

[a] Merged grasslands, pasture/hay, fallow, urban/recreational grasses, emergent herbaceous wetlands, grass-forb, and shrub-seedling forests ≥1.6 ha.

[b] Sites with a positive SI1 score (Table 11) and ≥ 0.3 ha.

and SI scores shows greater increases in suitability with relatively modest increases in diurnal habitat compared to the increases in suitability associated with similar proportional increases in openings.

Soil properties also influence American woodcock habitat suitability. This species feeds nearly exclusively on earthworms, which it probes for preferentially in moist loamy soils (Rabe and others 1983). Because soils with excessive clay or sand contain insufficient, accessible earthworms with which to support a foraging woodcock, we included both soil texture (SI4) and soil drainage (SI5) as variables in the habitat suitability model. We used the STATSGO database to define soil characteristics. Soil texture classes from STATSGO were crosswalked to soil texture classes from the soil triangle (Table 14) and then assigned SI scores on the basis of texture descriptions in Rabe and others (1983) (Table 15). We also assumed that soil drainage class was associated with soil moisture content and similarly assigned SI scores to these drainage classes (Table 16) based on observations from Rabe and others (1983), who documented higher probing rates in soils with greater moisture contents.

To calculate the overall HSI score, we determined the geometric mean of SI scores for forest structure (SI1 and SI2) and landscape factors (SI3, SI4 and SI5) separately and then the geometric mean of these means together.

$\text{Overall HSI} = ((\text{SI1} * \text{SI2})^{0.500} * (\text{SI3} * \text{SI4} * \text{SI5})^{0.333})^{0.500}$

**Table 14.—Crosswalk of soil texture classes defined in STATSGO soil database to soil texture triangle classes**

| STATSGO soil texture class | Soil texture triangle class |
|---|---|
| Clayey | Clay |
| Clayey over loamy | Clay |
| Clayey-skeletal | Clay |
| Coarse-loamy | Sandy loam |
| Coarse-silty | Sandy loam |
| Fine | Silt |
| Fine-loamy | Silt loam |
| Fine-loamy over clayey | Silty clay loam |
| Fine-loamy over sandy or sandy-skeletal | Silt loam |
| Fine-silty | Silt |
| Fine-silty over clayey | Silt |
| Loamy | Loam |
| Loamy-skeletal | Loam |
| Loamy-skeletal over clayey | Loam |
| Not used | None |
| Sandy | Sand |
| Very-fine | Silty clay |
| All others | None |

**Table 15.—Suitability index (SI) scores for American woodcock habitat based on soil texture triangle classes**

| Soil texture triangle class | SI score |
|---|---|
| Clay | 0.0[a] |
| Silty clay | 0.0[a] |
| Silty clay loam | 0.2[a] |
| Silt loam | 0.4[a] |
| Silt | 0.0[a] |
| Loam | 1.0[b] |
| Sandy loam | 0.8[b] |
| Loamy sands | 0.0[a] |
| Sands | 0.0[b] |
| Sandy clay loam | 0.4[a] |
| Sandy clay | 0.0[a] |
| Clay loam | 0.1[b] |
| None | 0.0[a] |

[a] Assumed value.
[b] Rabe and others (1983).

**Table 16.—Suitability index (SI) scores for American woodcock habitat based on soil moisture, as defined by drainage class in the STATSGO soil database**

| Soil moisture | SI score |
|---|---|
| Very poorly | 1.0[a] |
| Poorly | 1.0[a] |
| Somewhat poorly | 0.5[a] |
| Moderately well | 0.1[a] |
| Well | 0.0[a] |
| Somewhat excessively | 0.0[a] |
| Excessively | 0.0[a] |

[a] Rabe and others (1983).

## Verification and Validation

The American woodcock was observed only in 50 of the 88 subsections within the CH and WGCP. Spearman rank correlation identified a significant ($P \leq 0.001$) positive relationship ($r_s = 0.36$) between average HSI score and mean BBS route abundance across all subsections. When the 38 subsections in which the American woodcock was not found were removed from the analysis, the correlation not only remained significant ($P \leq 0.001$) but also was more strongly positive ($r_s = 0.68$). Thus, the HSI model is predicting habitat for this species in subsections where it was not detected on BBS routes. The generalized linear model predicting BBS abundance from BCR and HSI for the American woodcock was significant ($P \leq 0.001$; $R^2 = 0.218$), and the coefficient on the HSI predictor variable was both positive ($\beta = 0.090$) and significantly different from zero ($P \leq 0.001$). Therefore, we considered the HSI model for the American woodcock both verified and validated (Tirpak and others 2009a).

## Bachman's Sparrow


U.S. Forest Service

### Status

Bachman's sparrow (*Aimophila aestivalis*) is a resident bird associated with pine savannas and other open habitats throughout the Southeastern United States. Although its range expanded north to include Illinois, Indiana, and Ohio at the turn of the 20th century (likely in response to widespread land clearing), the range of this species has contracted steadily over the last 100 years. Today, the Bachman's sparrow is restricted to the extreme Southeast. BBS data from the central United States indicates significant annual declines (8.1 percent) over the past 40 years; declines have been particularly steep since 1980 (20.8 percent/year). This species is a Bird of Conservation Concern in both the CH and WGCP (Table 1). Similarly, this bird has a regional combined score of 20 in both regions, and PIF considers this species in need of critical recovery in the CH and immediate management in the WGCP (Table 1).

### Natural History

Bachman's sparrow occupies two primary habitats in the Southeast: mature (> 80 year old) pine stands that are frequently burned (< 3-year burn interval) and recently cutover areas (< 5 year old; Dunning and Watts 1990). However, productivity is lower in these latter habitats (one vs. three offspring/pair/year; Liu and others 1995, Perkins and others 2003a). On the basis of this lower productivity and the poor colonizing ability of this species—suitable clearcut habitats more than 3 km from a source population generally remained unoccupied in South Carolina (Dunning and others 1995)—Tucker and others (2004) considered Bachman's sparrow as endemic to mature longleaf pine stands.

In all studies of Bachman's sparrow habitat, two features are identified repeatedly: a dense grass understory and an open overstory, both of which are maintained through frequent fires (Haggerty 1998, Plentovich and others 1998, Tucker and others 2004, Wood and others 2004). Stands managed for the red-cockaded woodpecker via prescribed burning typically provide excellent habitat for the Bachman's sparrow as well because the fires are frequent enough to suppress dense woody understories and maintain sparse canopies (Wilson and others 1995, Plentovich and others 1998, Provencher and others 2002, Wood and others 2004).

### Model Description

Our habitat suitability model for the Bachman's sparrow includes six variables: landform, landcover type, successional age class, forest patch size, canopy cover, and connectivity.

The first suitability function combines landform, landcover type, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 17). We directly assigned SI scores to these combinations on the basis of data from Hamel (1992) on the relative quality of different vegetation types in different successional stages for this species.

**Table 17.—Relationship of landform, landcover type, and successional age class to suitability index scores for Bachman's sparrow habitat**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 1.000 | 0.333 | 0.000 | 0.000 | 1.000 |
| | Deciduous | 1.000 | 0.333 | 0.000 | 0.000 | 0.000 |
| | Evergreen | 1.000 | 0.333 | 0.000 | 0.000 | 1.000 |
| | Mixed | 1.000 | 0.333 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 1.000 | 0.333 | 0.000 | 0.000 | 1.000 |
| | Deciduous | 1.000 | 0.333 | 0.000 | 0.000 | 0.000 |
| | Evergreen | 1.000 | 0.333 | 0.000 | 0.000 | 1.000 |
| | Mixed | 1.000 | 0.333 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 1.000 | 0.333 | 0.000 | 0.000 | 1.000 |
| | Deciduous | 1.000 | 0.333 | 0.000 | 0.000 | 0.000 |
| | Evergreen | 1.000 | 0.333 | 0.000 | 0.000 | 1.000 |
| | Mixed | 1.000 | 0.333 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |

We also included forest patch size (SI2) as a variable because of the relatively large home range for this species (mean = 2.5 ha; Haggerty 1998). Home ranges varied among regions and habitat types (reviewed in Mitchell 1998). They were slightly larger in evergreen stands (4.8 ha) than in ephemeral, early successional habitats (2.2 ha). We fit a logistic function (Fig. 7) through these data points, assuming that the former represented a stand area that would be occupied reliably and that the latter value was a minimum below which the sparrow would be absent (Table 18).

We included canopy cover (SI3) as a third suitability function to satisfy the two-fold requirement for open canopies and dense understories, two habitat components often well correlated (Table 19). Haggerty (1998) observed an average canopy cover of 9.5 percent at sites occupied by the Bachman's sparrow and 40 percent canopy cover at unoccupied sites. Wood and others (2004) observed 20 times more Bachman's sparrows in habitats with 25 to 50 percent canopy cover than sites with 50 to 75 percent cover. We fit an inverse logistic function to these data to extrapolate values between these known points (Fig. 8).

Because this resident species is restricted to a specialized habitat, occupancy of a site by the Bachman's sparrow is affected by the ability of dispersers to colonize it. This ability is

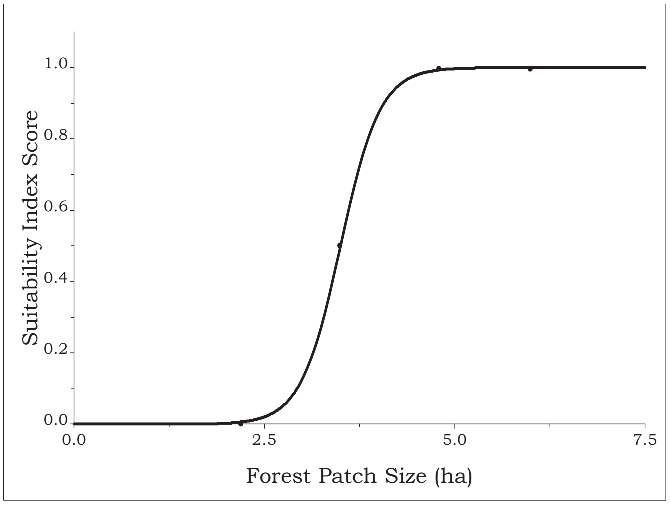


Figure 7.—Relationship between forest patch size and suitability index (SI) scores for Bachman's sparrow habitat. Equation: SI score = $1.000 / (1 + (699817.120 * e^{-3.845 * \text{forest patch size}}))$.

**Table 18.—Relationship between forest patch size and suitability index (SI) scores for Bachman's sparrow habitat**

| Forest patch size (ha) | SI score |
|---|---|
| 0.0[a] | 0.0 |
| 2.2[b] | 0.0 |
| 3.5[b] | 0.5 |
| 4.8[b] | 1.0 |
| 6.0[a] | 1.0 |

[a] Assumed value.

[b] Stober (1996), reviewed in Mitchell (1998).

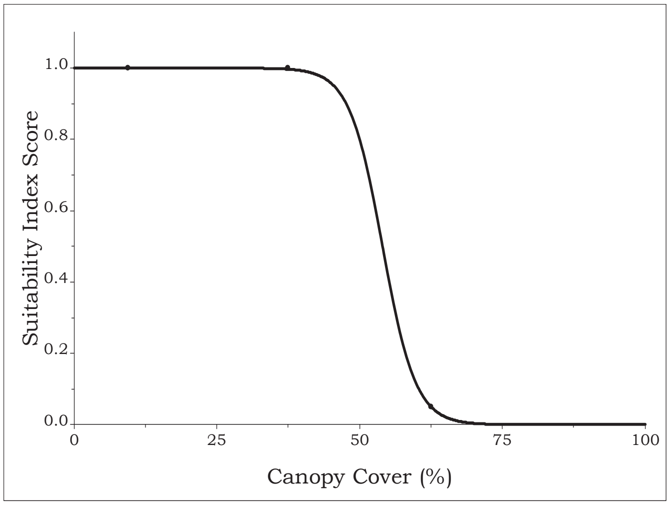


Figure 8.—Relationship between canopy cover and suitability index (SI) scores for Bachman's sparrow habitat. Equation: SI score = $1 - (1.000 / (1 + (126024970 * e^{-0.3455 * \text{canopy cover}})))$.

**Table 19.—Relationship between canopy cover and suitability index (SI) scores for Bachman's sparrow habitat**

| Canopy cover (percent) | SI score |
|---|---|
| 0.0[a] | 1.00 |
| 9.5[b] | 1.00 |
| 37.5[c] | 1.00 |
| 62.5[c] | 0.05 |
| 100.0[a] | 0.00 |

[a] Assumed value.

[b] Haggerty (1998).

[c] Wood and others (2004).

directly affected by the connectivity (or conversely the isolation) of habitat patches (SI4). Birds are unable to colonize clearcuts more than 3 km distant before succession renders habitat conditions within them unsuitable (Dunning and others 1995). Although isolation also may affect the occupancy of mature evergreen stands, habitat conditions within them are less ephemeral. Thus, the Bachman's sparrow has a potentially longer time to colonize these stands. To compensate for this differential temporal window in accessibility, we used a 15-km distance threshold to fit a longer tail to the function relating connectivity of patches to their suitability as Bachman's sparrow habitat (Table 20, Fig. 9). We also assumed that source populations were restricted to mature evergreen forest stands with a preliminary overall SI score (calculated from SI1, SI2, and SI3) that was greater than 0.8.

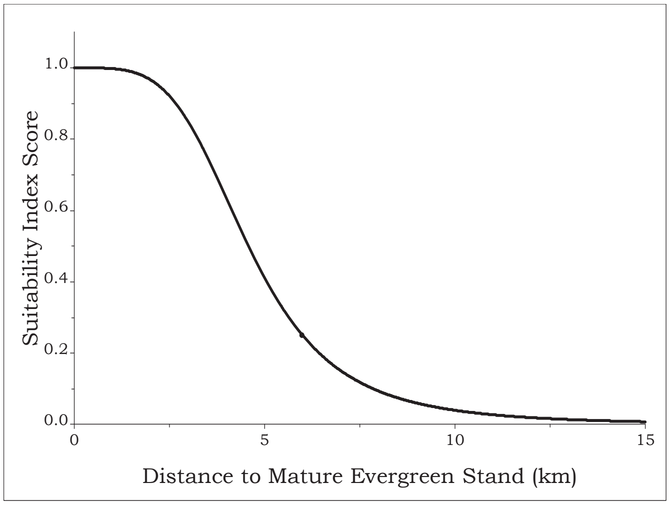


Figure 9.—Relationship between distance to nearest evergreen sawtimber habitat with initial suitability index (SI) score >0.8 and SI scores for Bachman's sparrow habitat. Equation: SI score = 1 / (1.000 + (0.002 * (distance to evergreen sawtimber habitat with initial SI score $>0.8)^{4.066}$)).

**Table 20.—Relationship between distance to nearest evergreen sawtimber habitat with initial suitability index (SI) score > 0.8 and SI scores for Bachman's sparrow habitat**

| Habitat connectivity (km) | SI score |
|---|---|
| 0[a] | 1.00 |
| 6[b] | 0.25 |
| 15[b] | 0.00 |

[a]Dunning and others (1995).
[b]Assumed value.

To calculate the overall HSI score, we calculated the geometric mean of the two SIs related to forest structure (SI1 and SI3) and landscape attributes (SI2 and SI4) separately and then the geometric mean of these values together.

$$\text{Overall HSI} = ((\text{SI1} * \text{SI3})^{0.500} * (\text{SI2} * \text{SI4})^{0.500})^{0.500}$$

## Verification and Validation

Bachman's sparrow was found only in 29 of the 88 subsections within the CH and WGCP. Spearman rank correlation identified a significant ($P \leq 0.001$) positive relationship ($r_s = 0.62$) between average HSI score and mean BBS route abundance across all subsections. However, when subsections where the Bachman's sparrow was not found were removed from the analysis, the relationship was not significant ($r_s = 0.24$; P = 0.208). Thus, the HSI model predicts the absence of the Bachman's sparrow better than its abundance in subsections where it is found. The generalized linear model predicting BBS abundance from BCR and HSI for the Bachman's sparrow was significant ($P \leq 0.001$; $R^2 = 0.567$), and the coefficient on the HSI predictor variable was both positive ($\beta = 0.908$) and significantly different from zero ($P = 0.079$). Therefore, we considered the HSI model for the Bachman's sparrow both verified and validated (Tirpak and others 2009a).

# Bell's Vireo

## Status

Steve Maslowski, U.S. Fish & Wildlife Service

Bell's vireo (*Vireo bellii*) is a scrubland specialist that reaches the eastern limit of its range in the CH and WGCP. Throughout both regions this species has declined over the past 40 years, with the most severe declines in the southern portion of the eastern range (-4.7, -6.6, and -10.1 percent annually in Missouri, Oklahoma, and the Ozark-Ouachita Plateau, respectively; Sauer and others 2005). Bell's vireo has a regional combined score of 15 in the CH and 16 in the WGCP, and PIF considers the species as requiring management attention in both regions (Table 1). The FWS also recognizes Bell's vireo as a Bird of Conservation Concern in both BCRs (Table 1).

## Natural History

Bell's vireo is a small, Neotropical migrant associated with dense, low, shrubby vegetation (Brown 1993). It uses a variety of early successional scrubland habitats that meet these requirements (e.g., riparian woods, brushy fields, and regenerating forest). Most of the research on this species was conducted in the West, where Bell's vireo is alternately described as a riparian specialist (particularly the federally endangered subpopulation of least Bell's vireo in California) or a scrub-shrub generalist. This bird nests in dense shrub or understory vegetation 0.5 to 1.5 m above the ground, making its nests susceptible to both terrestrial and avian predators. Predation and brood parasitism are the primary causes of nest failure (Budnik and others 2000, 2002; Powell and Steidl 2000). Increasing the density of large shrub patches may improve Bell's vireo habitat in Missouri (Budnik and others 2002).

## Model Description

The model for Bell's vireo includes six variables: landform, landcover, successional age class, interspersion of forest and open areas, edge, and small stem density.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 21). We directly assigned SI values to these combinations on the basis of data from Hamel (1992) relating vegetation types and successional age class to habitat suitability estimates for Bell's vireo.

Both landcover and age class data were used to identify upland shrublands in grassland landscapes, the preferred habitat for this species in its eastern range (Budnik and others 2000). We used a 10-ha moving window (an average home range; Budnik and others 2000) to assess the interspersion of shrubland and grassland habitats (SI2). We assumed that an area containing 50 percent of each habitat type was ideal (Table 22). To extrapolate from this point we used broad incremental changes in habitat suitability (20 percent) and applied these symmetrically to 10-percent incremental changes in the proportion of scrubland or grassland. Landscapes lacking shrublands or grasslands were unsuitable and assigned an SI score of zero.

**Table 21.—Relationship of landform, landcover type, and successional age class to suitability index scores for Bell's vireo habitat**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.500 | 0.250 | 0.125 | 0.000 | 0.000 |
| | Deciduous | 0.500 | 0.250 | 0.125 | 0.000 | 0.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.500 | 0.250 | 0.125 | 0.000 | 0.000 |
| | Woody wetlands | 0.500 | 0.500 | 0.250 | 0.000 | 0.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.500 | 1.000 | 0.750 | 0.000 | 0.000 |
| | Deciduous | 0.250 | 0.500 | 0.375 | 0.000 | 0.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.250 | 0.500 | 0.375 | 0.000 | 0.000 |
| | Woody wetlands | 1.000 | 0.500 | 0.250 | 0.000 | 0.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.500 | 1.000 | 0.750 | 0.000 | 0.000 |
| | Deciduous | 0.500 | 1.000 | 0.750 | 0.000 | 0.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.500 | 1.000 | 0.750 | 0.000 | 0.000 |
| | Woody wetlands | 1.000 | 0.500 | 0.250 | 0.000 | 0.000 |

**Table 22.—Relative composition of scrubland and grassland within 10-ha moving window on suitability index scores for Bell's vireo habitat**

| | Proportion grassland[a] | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Proportion scrubland[b] | 0.0 | 0.1 | 0.2 | 0.3 | 0.4 | 0.5 | 0.6 | 0.7 | 0.8 | 0.9 | 1.0 |
| 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| 0.1 | 0.0 | 0.0 | 0.0 | 0.1 | 0.2 | 0.2 | 0.2 | 0.2 | 0.2 | 0.2 | |
| 0.2 | 0.0 | 0.0 | 0.1 | 0.2 | 0.4 | 0.4 | 0.4 | 0.4 | 0.4 | | |
| 0.3 | 0.0 | 0.1 | 0.2 | 0.4 | 0.6 | 0.6 | 0.6 | 0.6 | | | |
| 0.4 | 0.0 | 0.2 | 0.4 | 0.6 | 0.8 | 0.8 | 0.8 | | | | |
| 0.5 | 0.0 | 0.2 | 0.4 | 0.6 | 0.8 | 1.0[c] | | | | | |
| 0.6 | 0.0 | 0.2 | 0.4 | 0.6 | 0.8 | | | | | | |
| 0.7 | 0.0 | 0.2 | 0.4 | 0.6 | | | | | | | |
| 0.8 | 0.0 | 0.2 | 0.4 | | | | | | | | |
| 0.9 | 0.0 | 0.2 | | | | | | | | | |
| 1.0 | 0.0 | | | | | | | | | | |

[a] Grasslands/herbaceous, pasture/hay, and grass-forb successional age class.

[b] Shrub-seedling and sapling successional age classes.

[c] Budnik and others (2000); all other values assumed.

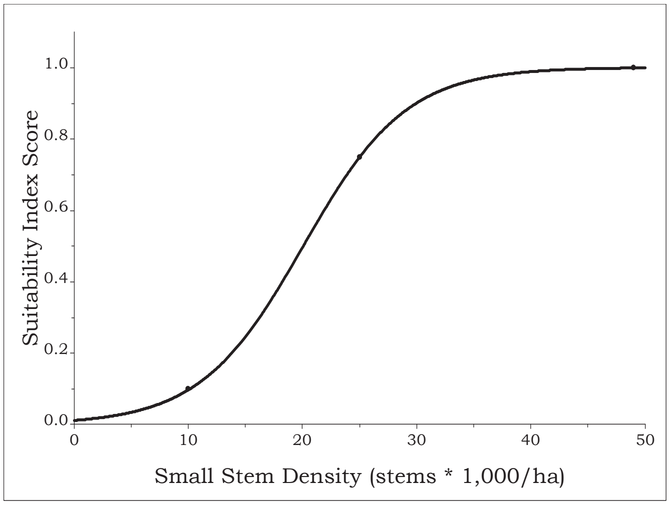


Figure 10.—Relationship between small stem (< 2.5 cm d.b.h.) density (stems * 1000/ha) and suitability index (SI) scores for Bell's vireo habitat. Equation: SI score = 1.001 / (1.000 + (85.005 * e $^{-0.222 * (\text{small stem density} / 1000)}$)).

**Table 23.—Influence of edge occurrence on suitability index (SI) scores for Bell's vireo habitat**

| 3 × 3 pixel window around forest pixel includes field[a] | SI score |
|---|---|
| Yes[b] | 1.0 |
| No | 0.0 |

[a] Field defined as any shrub-seedling or grass-forb age class pixel, natural grasslands/herbaceous, or pasture/hay. Forest defined as any used sapling age class pixel of transitional, shrublands, deciduous, orchard, or woody wetlands.

[b] Grass-forb and seedling-shrub habitats used regardless of edge.

**Table 24.—Relationship between small stem (< 2.5 cm d.b.h.) density (stems * 1,000/ha) and suitability index (SI) scores for Bell's vireo habitat**

| Small stem density | SI score |
|---|---|
| 0[a] | 0.00 |
| 10[a] | 0.10 |
| 25[a] | 0.75 |
| 49[b] | 1.00 |

[a] Assumed value.

[b] Farley (1987).

Bell's vireo uses a variety of young woody habitats (Brown 1993); however, birds also nest along the edges of sapling stands and in hedgerows (Budnik and others 2002). Therefore, we included edge (SI3) as a parameter in the Bell's vireo HSI model. To identify edges, we examined the eight pixels surrounding each sapling age class pixel to determine whether any were classified as shrub-seedling or grass-forb age class forest or as a nonforest landcover class. If so, the central pixel in the 3 × 3 pixel window (90 x 90 m) was assigned an SI score of 1.000; if not, it was assigned a zero. We assigned to grass-forb and shrub-seedling pixels an SI score of 1.000 regardless of edge (Table 23). Similarly, we always assigned to pole and sawtimber pixels an SI score of zero regardless of edge.

We also included small stem density (SI4) as a component of the overall Bell's vireo HSI model because of the importance of dense woody shrub cover for this species. Farley (1987) measured an average of 9.8 stems greater than 2 mm per 1-m diameter plot (approximately 392,000 stems/ha) in Bell's vireo territories. This relatively high stem value included woody and nonwoody stems of all sizes greater than 2 mm; therefore, we assumed that that only one-eighth of these stems (49,000 = ⅛ * 392,000) were woody and less than 2.5 cm d.b.h. and that this value represented optimal habitat (Table 24, Fig. 10).

To calculate the overall HSI score for Bell's vireo, we first determined the geometric mean of the suitability indices related to forest structure (SI1 and SI4) and landscape attributes (SI2 and SI3) separately and then determined the geometric mean of these values together. Because SI3 applies only to sapling habitats, HSI scores were calculated differently for sapling successional age class stands than for grass-forb or shrub-seedling successional age class stands. To determine the overall SI score across the entire BCR, we added suitability scores from individual age classes across the entire landscape.

For grass-forb and shrub-seedling habitats:

$$HSI_{\text{GF and SS}} = (((SI1 * SI4)^{0.500}) * (SI2))^{0.500}$$

For sapling habitats:

$$HSI_{\text{Sap}} = ((SI1 * SI4)^{0.500} * (SI2 * SI3)^{0.500})^{0.500}$$

$$\text{Overall HSI} = HSI_{\text{GF and SS}} + HSI_{\text{Sap}}$$

## Verification and Validation

Bell's vireo was found in 54 of the 88 subsections within the CH and WGCP. Spearman rank correlation identified a significant ($P \leq 0.001$) positive relationship ($r_s = 0.44$) between average HSI score and mean BBS route abundance across all subsections. Removing subsections in which Bell's vireo was not observed had a minimal effect on these results ($r_s = 0.46$; $P \leq 0.001$). The generalized linear model predicting BBS abundance from BCR and HSI for the Bell's vireo was significant ($P = 0.042$; $R^2 = 0.072$); however, the coefficient on the HSI predictor variable was negative ($\beta = -19.906$) and not significantly different from zero ($P = 0.544$). Therefore, we considered the HSI model for the Bell's vireo verified but not validated (Tirpak and others 2009a).

## Bewick's Wren

Dave Menke, U.S. Fish & Wildlife Service

### Status

Bewick's wren (*Thryomanes bewickii*) was once a common resident throughout the Southeast and mid-Atlantic. However, its range has contracted steadily over the last century and today this species is virtually absent east of the Mississippi River (Kennedy and White 1997). BBS data from FWS Region 4 indicates that populations have declined by 12.8 percent per year over the last 40 years (Sauer and others 2005). The decline of this species coincided with the range expansion of the house wren, which often destroys Bewick's wren nests in areas where the species' ranges overlap (Kennedy and White 1996). Bewick's wren is a Bird of Conservation Concern in both the CH and WGCP (Table 1). PIF identifies the species as requiring both critical recovery in the WGCP (regional combined score = 16) and immediate management attention in the CH (regional combined score = 15).

### Natural History

Bewick's wren is a small resident passerine that breeds in a variety of vegetation types, including brushy areas, scrub and thickets in open country, and open and riparian woodlands (Kennedy and White 1997). This plasticity has produced conflicting reports of habitat associations in the literature (e.g., dry vs. riparian, open woodlands vs. shrub thickets). However, this species likely responds most strongly to the availability of nest sites. Bewick's wren nests in cavities or opportunistically in crevices up to 10 m high. In the eastern portion of its range, this bird often lives near human habitation, particularly farmland. As mentioned, population declines of this species may be partly the result of competition with the house wren (Kennedy and White 1996). Bewick's wren is found primarily in grassland scrub while the house wren occurs primarily in secondary growth on abandoned agricultural land and in residential areas. Both species exploit the full range of these habitat types, and populations of both expanded as these latter types increased. However, as scrub habitats declined, Bewick's wren may have declined because its primary source habitat no longer was abundant.

### Model Description

Our model for Bewick's wren includes five variables: landform, landcover, successional age class, interspersion of forest and open habitats, and snag density.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 25). We then directly assigned an SI score to these combinations on the basis of data from Hamel (1992) on the relative quality of Bewick's wren habitat based on vegetation type and successional age class.

We also considered as important for this species the interspersion of forest and grassland habitats (SI2), as Bewick's wren is most abundant in semi-open areas containing about 40 percent woodland (Pogue and Schnell 1994; Table 26). We relied on data from Pogue and Schnell to define SI values along the diagonal axis of our interspersion table (where forest and grassland totaled 100 percent) and completed the table from these values.

**Table 25.—Relationship of landform, landcover type, and successional age class to suitability index scores for Bewick's wren habitat**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 1.000 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Transitional-shrubland | 1.000 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Deciduous | 0.500 | 0.500 | 0.250 | 0.000 | 0.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 1.000 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Terrace-mesic | Low-density residential | 1.000 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Transitional-shrubland | 1.000 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Deciduous | 0.500 | 0.500 | 0.250 | 0.000 | 0.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 1.000 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Xeric-ridge | Low-density residential | 1.000 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Transitional-shrubland | 1.000 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Deciduous | 0.500 | 0.500 | 0.250 | 0.000 | 0.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 1.000 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |

**Table 26.—Influence of interspersion between forest and open habitats (as indexed by relative composition within 10-ha moving window) on suitability index scores for Bewick's wren habitat**

| | Proportion open[a] | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Proportion forest[b] | 0.0 | 0.1 | 0.2 | 0.3 | 0.4 | 0.5 | 0.6 | 0.7 | 0.8 | 0.9 | 1.0 |
| 0.0 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00[c] |
| 0.1 | 0.00 | 0.00 | 0.05 | 0.10 | 0.10 | 0.20 | 0.20 | 0.20 | 0.20 | 0.20[c] | |
| 0.2 | 0.00 | 0.05 | 0.10 | 0.15 | 0.20 | 0.25 | 0.40 | 0.40 | 0.40[c] | | |
| 0.3 | 0.00 | 0.05 | 0.20 | 0.25 | 0.60 | 0.80 | 0.80 | 0.80[c] | | | |
| 0.4 | 0.00 | 0.05 | 0.20 | 0.40 | 0.80 | 1.00 | 1.00[c] | | | | |
| 0.5 | 0.00 | 0.05 | 0.20 | 0.40 | 0.80 | 1.00[c] | | | | | |
| 0.6 | 0.00 | 0.10 | 0.20 | 0.40 | 0.80[c] | | | | | | |
| 0.7 | 0.00 | 0.10 | 0.20 | 0.40[c] | | | | | | | |
| 0.8 | 0.00 | 0.10 | 0.20[c] | | | | | | | | |
| 0.9 | 0.00 | 0.10[c] | | | | | | | | | |
| 1.0 | 0.00[c] | | | | | | | | | | |

[a] Open = grasslands, herbaceous planted (pasture-hay, fallow, and urban-recreational grasses), emergent herbaceous wetlands.

[b] Forest = forested upland, low-density residential, shrubland, transitional, and woody wetlands.

[c] Pogue and Schnell (1994); all other values assumed.

**Table 27.—Influence of snag density on suitability index scores for Bewick's wren habitat**

| Snag density (snags/ha) | SI score |
|---|---|
| 6.2[a] | 0.128 |
| 16.4[b] | 0.500 |
| 52.8[a] | 1.000 |

[a] Rumble and Gobeille (2004).
[b] Sedgwick and Knopf (1990).

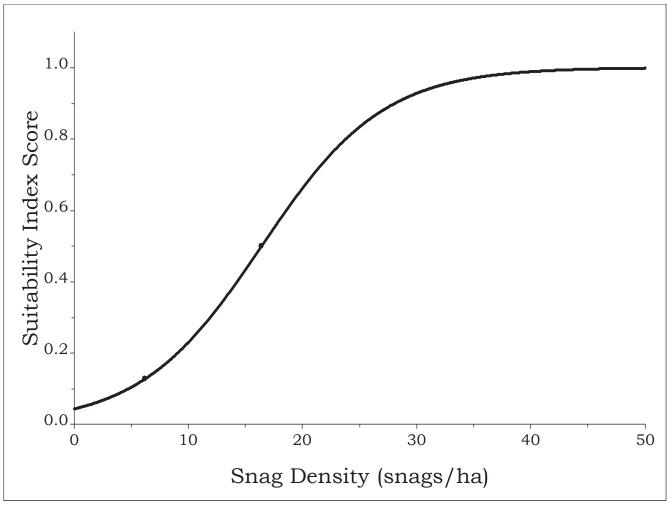


Figure 11.—Relationship between snag density and suitability index (SI) score for Bewick's wren habitat. Equation: SI score = 1.0011 / (1 + (21.9129 * e $^{-0.1881 * \text{snag density}}$)).

We also included snag density (SI3) in our model of Bewick's wren habitat because as a secondary cavity nester, this species responds strongly to nest-site availability. We assumed that higher snag densities would decrease competition with other cavity nesters, improving habitat quality. Specific data relating snag density to Bewick's wren habitat suitability were not available, so we assumed that the average snag density observed by Sedgwick and Knopf (1990) (16.4 snags/ha) within home ranges of the house wren, a secondary cavity nester of similar size, represented average habitat suitability (SI score = 0.500) for the Bewick's wren. We coupled this information with data from Rumble and Gobeille (2004) (Table 27) on the relative density of the house wren in habitats with different snag densities to build a logistic function quantifying the relationship between habitat suitability and snag density (Fig. 11).

To calculate the overall HSI score, we first calculated the geometric mean of the two suitability indices related to forest structure attributes (SI1 and SI3), and then the geometric mean of this result and the SI related to interspersion (SI2).

Overall HSI = $((\text{SI1} * \text{SI3})^{0.500} * \text{SI2})^{0.500}$

## Verification and Validation

Bewick's wren was found in 74 of the 88 subsections within the CH and WGCP. Spearman rank correlation identified a significant ($P \leq 0.001$) positive relationship ($r_s = 0.40$) between average HSI score and mean BBS route abundance across subsections. However, this relationship was weaker ($r_s = 0.35$; $P = 0.002$) when subsections in which the Bewick's wren was not detected were removed from the analysis. The generalized linear model predicting BBS abundance from BCR and HSI for the Bewick's wren was not significant ($P = 0.517$; $R^2 = 0.015$), and the coefficient on the HSI predictor variable was negative ($\beta = -3.193$) and not significantly different from zero ($P = 0.857$). Therefore, we considered the HSI model for the Bewick's wren verified but not validated (Tirpak and others 2009a).

## Black-and-white Warbler

Charles H. Warren, images.nbii.gov

### Status

The black-and-white warbler (*Mniotilta varia*) is a neotropical migrant found throughout the eastern United States and southern Canada. This is a forest-interior species and the annual declines of 1.2 percent observed in the United States over the last 40 years likely are the result of increasing forest fragmentation (Sauer and others 2005). This species has a regional combined score of 16 in the WGCP, where it is a species requiring management attention (Table 1). The black-and-white warbler has a regional combined score of only 13 in the CH. The FWS does not recognize the black-and-white warbler as a Bird of Conservation Concern in either BCR (Table 1).

### Natural History

As a forest-interior specialist, the black-and-white warbler is found in the mature deciduous hardwood forests of the eastern United States and Canada (Kricher 1995). It is highly sensitive to fragmentation in the landscape (Robbins and others 1989) and typically is absent from small woodlots (< 7.5 ha; Galli and others 1976). Hamel (1992) suggested that 550 ha was the minimum tract size for this species in the Southeast.

Few studies have focused exclusively on the habitat ecology of this bird, though Conner and others (1983) found that the black-and-white warbler is associated with mature forest stands with high densities of large (> 32 cm d.b.h.) trees. Although a ground-nesting bird, this species is associated with high densities of hardwood saplings. Conversely, pine saplings negatively affect both the presence and abundance of the black-and-white warbler.

This bird occupies upland and bottomland forests but reaches greater densities in the former, with oak-hickory and cove forests considered optimal (Hamel 1992). Nevertheless, successional age may be the most critical habitat factor affecting the black-and-white warbler. Dettmers and others (2002) validated Hamel's (1992) habitat suitability model for the black-and-white warbler, finding the model performed well due to the restriction of the black-and-white warbler to older age class forests. However, Thompson and others (1992) and Annand and Thompson (1997) observed the black-and-white warbler in sapling and clearcut stands in Missouri.

### Model Description

Our HSI model for the black-and-white warbler includes six variables: landform, landcover, successional age class, forest patch size, percent forest in a 1-km radius, and canopy cover.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 28). We directly assigned SI scores to these combinations based on vegetation type and age class associations of the black-and-white warbler reported by Hamel (1992). However, we assigned higher values to shrub-seedling stands based on data from Thompson and others (1992) and Annand and Thompson (1997).

**Table 28.—Relationship between landform, landcover type, age class, and suitability index scores for black-and-white warbler habitat; values in parentheses apply to West Gulf Coastal Plain/Ouachitas**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.167 | 0.333 | 0.333 | 0.667 |
| | Deciduous | 0.000 | 0.167 | 0.333 | 0.333 | 0.667 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.167 | 0.333 | 0.333 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.167 | 0.333 | 0.333 | 0.333 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.167<br>(0.000) | 0.333<br>(0.000) | 0.333<br>(0.000) | 0.333<br>(0.000) |
| | Deciduous | 0.000 | 0.167 | 0.333 | 0.333 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.167 | 0.333 | 0.333 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.167 | 0.333 | 0.333 | 0.333 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.167<br>(0.000) | 0.333<br>(0.000) | 0.333<br>(0.000) | 0.333<br>(0.000) |
| | Deciduous | 0.000 | 0.167 | 0.333 | 0.333 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.167 | 0.333 | 0.333 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.167 | 0.333 | 0.333 | 0.333 |

Forest patch size (SI2) affects occurrence of this species as it is notably absent from small forest blocks. Therefore, we fit a logarithmic function (Fig. 12) relating forest patch size to SI scores derived from probability of occurrence data from Robbins and others (1989) (Table 29). The relative value of a forest block of a specific size is influenced by its landscape context. In predominantly forested landscapes, small forest patches that may not be used in predominantly nonforested landscapes may provide habitat due to their proximity to large forest blocks (Rosenberg and others 1999). To capture this relationship, we fit a logistic function (Fig. 13) to data (Table 30) derived from Donovan and others (1997), who observed differences in predator and brood parasite communities among highly fragmented (< 15 percent), moderately fragmented (45 to 50 percent), and lightly fragmented (> 90 percent forest) landscapes. Because of the extreme sensitivity of the black-and-white warbler to fragmented landscapes, we assumed that the midpoint between moderately and lightly fragmented forest defined the specific cutoff for average (SI score = 0.500) hatitat. We used the maximum value of SI2 or SI3 to account for small patches in predominantly forested landscapes and large patches in predominantly nonforested landscapes.

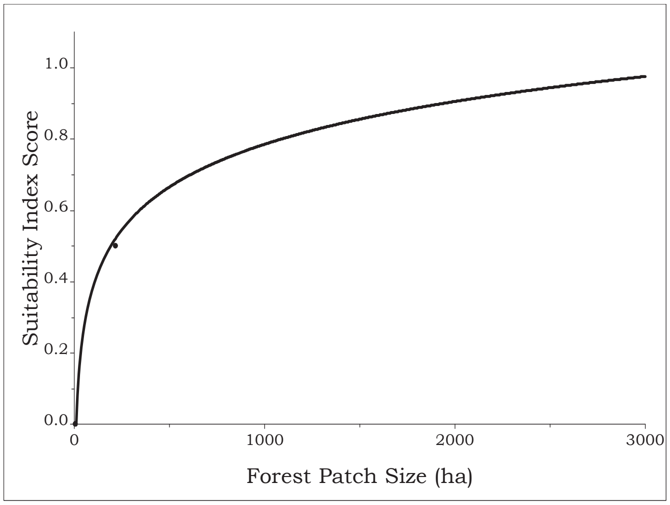


Figure 12.—Relationship between forest patch size and suitability index (SI) scores for black-and-white warbler habitat. Equation: SI score = 0.1731 * ln(forest patch size) – 0.4096.

**Table 29.—Influence of forest patch size on suitability index (SI) scores for black-and-white warbler habitat**

| Forest patch size (ha) | SI score |
|---|---|
| 10[a] | 0.0 |
| 220[b] | 0.5 |
| 3,200[b] | 1.0 |

[a] Assumed value.

[b] Robbins and others (1989).

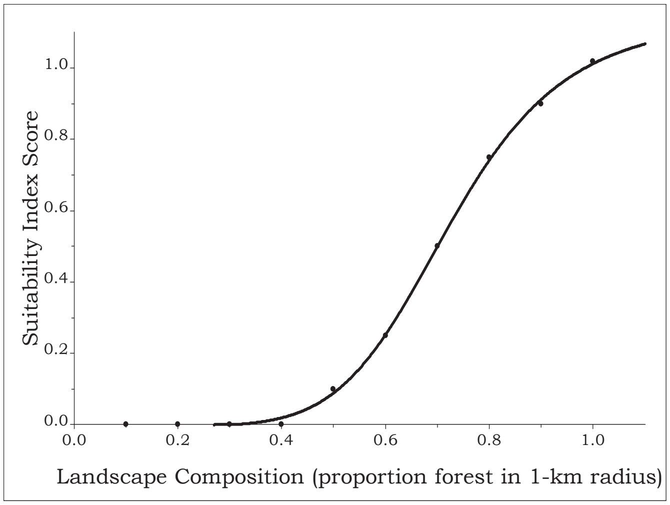


Figure 13.—Relationship between landscape composition and suitability index (SI) scores for black-and-white warbler habitat. Equation: SI score = $1.047 / (1.000 + (1991.516 * e^{-10.673 * \text{landscape composition}}))$.

**Table 30.—Relationship between landscape composition (proportion forest in 1-km radius) and suitability index (SI) scores for black-and-white warbler habitat**

| Landscape composition[a] | SI score |
|---|---|
| 0.00[a] | 0.00 |
| 0.10[a] | 0.00 |
| 0.20[a] | 0.00 |
| 0.30[a] | 0.00 |
| 0.40[a] | 0.00 |
| 0.50[a] | 0.10 |
| 0.60[a] | 0.25 |
| 0.70[b] | 0.50 |
| 0.80[a] | 0.75 |
| 0.90[a] | 0.90 |
| 1.00[a] | 1.00 |

[a] Assumed value.

[b] Donovan and others (1997).

Canopy cover (SI4) also may affect the quality of black-and-white warbler habitat. Thus, we included it as a factor in our HSI model. Prather and Smith (2003) reported higher densities of the black-and-white warbler in forests with relatively open canopies, so we used their data (Table 31) to derive an inverse logistic function (Fig. 14) that quantified the relationship between canopy cover and SI scores.

We calculated the overall HSI score as the geometric mean of the geometric mean of individual SI functions related to forest structure (SI1 and SI4) multiplied by the maximum SI score for forest patch size or percent forest in the 1-km radius landscape.

Overall HSI = $((\text{SI1} * \text{SI4})^{0.500} * \text{Max}(\text{SI2 or SI3}))^{0.500}$

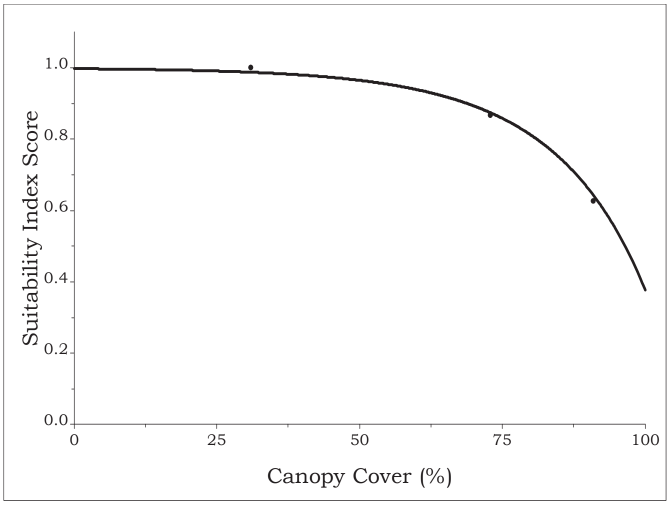


Figure 14.—Relationship between canopy cover and suitability index (SI) scores for black-and-white warbler habitat. Equation: SI score = 1 - (-4.190 / (1 + (-1890.213 * $e^{-0.055 * \text{canopy cover}}$))).

**Table 31.—Influence of canopy cover on suitability index (SI) scores for black-and-white warbler habitat.**

| Canopy cover (percent)[a] | SI score |
|---|---|
| 31 | 1.000 |
| 73 | 0.866 |
| 91 | 0.627 |

[a] Prather and Smith (2003).

## Verification and Validation

The black-and-white warbler was found in 85 of the 88 subsections within the CH and WGCP. Not surprisingly, Spearman rank correlations based on all subsections and only subsections in which this species was found produced similar results: significant ($P \leq 0.001$ for both analyses) positive relationships ($r_s = 0.54$ and 0.53, respectively) between average HSI score and mean BBS route abundance. The generalized linear model predicting BBS abundance from BCR and HSI for the black-and-white warbler was significant ($P \leq 0.001$; $R^2 = 0.380$), and the coefficient on the HSI predictor variable was both positive ($\beta = 3.194$) and significantly different from zero ($P \leq 0.001$). Therefore, we considered the HSI model for the black-and-white warbler both verified and validated (Tirpak and others 2009a).

## Blue-gray Gnatcatcher

Charles H. Warren, images.nbii.gov

### Status

The blue-gray gnatcatcher (*Polioptila caerulea*) is a short-distance migrant found throughout eastern North America and the Southwest. Populations are relatively stable in both the CH and WGCP (Table 5). The FWS does not recognize this species as a Bird of Conservation Concern in either region (Table 1). This bird requires management attention in the CH (regional combined score = 14) but does not have any special designation in the WGCP (regional combined score =13; Table 1).

### Natural History

The blue-gray gnatcatcher is a small passerine that inhabits woodland types ranging from shrubland to mature forest (Ellison 1992). It prefers deciduous habitats and is rare or absent in evergreen forests. This species attains its highest numbers in mesic and low-lying areas, but is also found in xeric forests and along ridges.

Kershner and others (2001) did not identify specific microhabitat requirements for this species in Illinois, and considerable variation in nest height (0.8 to 24.4 m) and territory size (0.5 to 8 ha) has been documented across the range.

Although often associated with edges, this bird may be area sensitive (Knutson 1995, Kilgo and others 1998). Nest success was greater for nests placed higher and farther from an edge in Illinois (Kershner and others 2001) but did not differ between bottomland hardwood stands and cottonwood plantations in the Mississippi Alluvial Valley (Twedt and others 2001). The abundance of the blue-gray gnatcatcher was higher in bottomland hardwood stands surrounded by fields than those surrounded by pine forest (Kilgo and others 1998).

### Model Description

The HSI model for the blue-gray gnatcatcher includes seven variables in five functions: landform, landcover, successional age class, forest patch size, percent forest in a 1-km radius landscape, edge, and basal area.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 32). We directly assigned SI scores to these combinations on the basis of data from Hamel (1992) on the relative quality of vegetation associations and successional age classes for this species. We adjusted Hamel's values for shrub-seedling and sapling-aged stands to account for the higher densities observed in young forests by Thompson and others (1992) and Annand and Thompson (1997).

We included forest patch size (SI2) as a variable to account for the area sensitivity of the blue-gray gnatcatcher. We fit a logarithmic function (Fig. 15) to data from Robbins and others (1989) on the probability of occurrence for this bird in stands of various sizes (Table 33). Nevertheless, the actual use of a forest patch reflects both its area and its landscape

**Table 32.—Relationship of landform, landcover type, and successional age class to suitability index scores for blue-gray gnatcatcher habitat; values in parentheses apply to West Gulf Coastal Plain/Ouachitas**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.333 | 0.667 | 0.667 | 1.000 |
| | Transitional-shrubland | 0.000 | 0.333 | 0.667 | 0.667 | 1.000 |
| | Deciduous | 0.000 | 0.333 | 0.667 | 0.667 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.083 | 0.167 | 0.167 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.167 | 0.333 | 0.333 | 1.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.333 | 0.667 | 0.667 | 1.000 |
| | Transitional-shrubland | 0.000 | 0.333<br>(0.000) | 0.667<br>(0.000) | 0.667<br>(0.000) | 1.000<br>(0.000) |
| | Deciduous | 0.000 | 0.333 | 0.667 | 0.667 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.083 | 0.167 | 0.167 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.167 | 0.333 | 0.333 | 1.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.333 | 0.667 | 0.667 | 1.000 |
| | Transitional-shrubland | 0.000 | 0.333<br>(0.000) | 0.667<br>(0.000) | 0.667<br>(0.000) | 1.000<br>(0.000) |
| | Deciduous | 0.000 | 0.333 | 0.667 | 0.667 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.083 | 0.167 | 0.167 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.167 | 0.333 | 0.333 | 1.000 |

context (SI3). In predominantly forested landscapes, a small forest patch that otherwise may not be suitable may be occupied due to its proximity to a larger forest block (Rosenberg and others 1999). Because the gnatcatcher also is associated with edges, it may not be as abundant in predominantly forested landscapes that lack significant edge habitat. Thus, we assumed that the relationship between habitat suitability of the blue gray gnatcatcher and the amount of forest in the landscape followed a Gaussian function (Fig. 16), with landscapes containing 70 to 80 percent forest as optimal and suitability declining as the proportion of forest in the landscape moved from this ideal (Table 34). We used the maximum suitability score of SI2 or SI3 to simultaneously account for patch area and landscape composition.

We also included edge (SI4) in our HSI model because of the association of the blue-gray gnatcatcher with edges within large forest blocks. This species nests along both hard and soft edges (typically within 30 m; Kershner and others 2001). Therefore, we defined edge as the interface among sapling, pole, and sawtimber stands and herbaceous and nonforest landcovers (hard edge) or seedling and grass-forb stands (soft edge). We used a 7 × 7 pixel moving window (210 x 210 m) to identify where these adjacencies occurred but recognized that the blue-gray gnatcatcher is not restricted to edge habitats and applied a residual SI score (0.010) to sites that did not meet this criterion (Table 35).

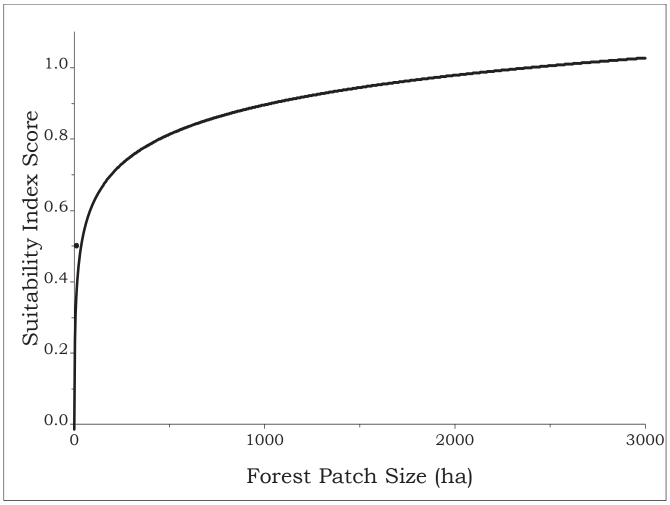


Figure 15.—Relationship between forest patch size and suitability index (SI) scores for blue-gray gnatcatcher habitat. Equation: SI score = 0.137 * ln(forest patch size) + 0.186.

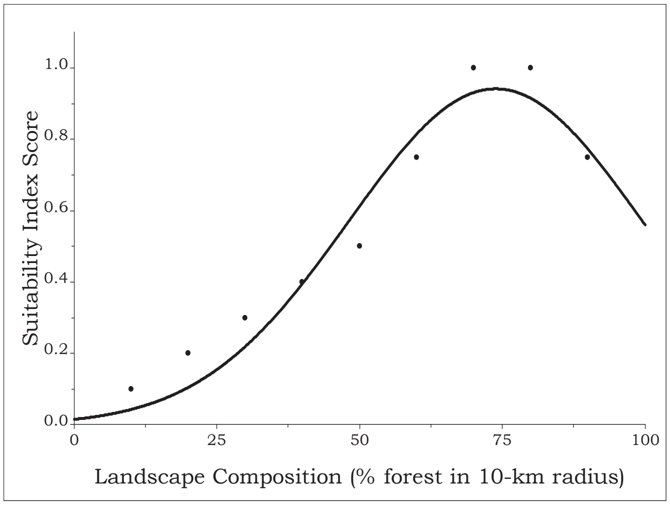


Figure 16.—Relationship between landscape composition and suitability index (SI) scores for blue-gray gnatcatcher habitat. Equation: SI score = 1.002 * e $^{((0 – ((\text{landscape composition}) – 74.165) \wedge 2) / 1064.634)}$.

**Table 33.—Influence of forest patch size on suitability index (SI) scores for blue-gray gnatcatcher habitat**

| Forest patch size (ha)[a] | SI score |
|---|---|
| 6.8 | 0.0 |
| 15 | 0.5 |
| 3,200 | 1.0 |

[a] Robbins and others (1989).

**Table 34.—Relationship between landscape composition (percent forest in 10-km radius) and suitability index (SI) scores for blue-gray gnatcatcher habitat**

| Landscape composition | SI score |
|---|---|
| 0[a] | 0.00 |
| 10[a] | 0.10 |
| 20[a] | 0.20 |
| 30[b] | 0.30 |
| 40[a] | 0.40 |
| 50[b] | 0.50 |
| 60[a] | 0.75 |
| 70[b] | 1.00 |
| 80[a] | 1.00 |
| 90[b] | 0.75 |
| 100[a] | 0.50 |

[a] Assumed value.

[b] Dononvan and others (1997).

**Table 35.—Influence of edge on suitability index (SI) scores for blue-gray gnatcatcher habitat**

| 7 × 7 pixel window around forest pixel includes field[a] | SI score |
|---|---|
| Yes | 1.00 |
| No | 0.01 |

[a] Field defined as any shrub-seedling or grass-forb age class forest, or natural grasslands, pasture-hay, fallow, urban-recreational grasses, emergent herbaceous wetlands, open water, high intensity residential, commercial-industrial-transportation, bare rock-sand-clay, quarries-strip mines-gravel pits, row crops, or small grains. Forest defined as any used sapling, pole, or sawtimber age class pixel of low-density residential, transitional, shrublands, deciduous, mixed, evergreen, orchard, or woody wetlands (i.e., SI1 > 0).

We fit a quadratic function to data from Annand and Thompson (1997) on the response of the blue-gray gnatcatcher to basal area (SI5; Table 36, Fig. 17), reflecting the preference of this species for open forest conditions.

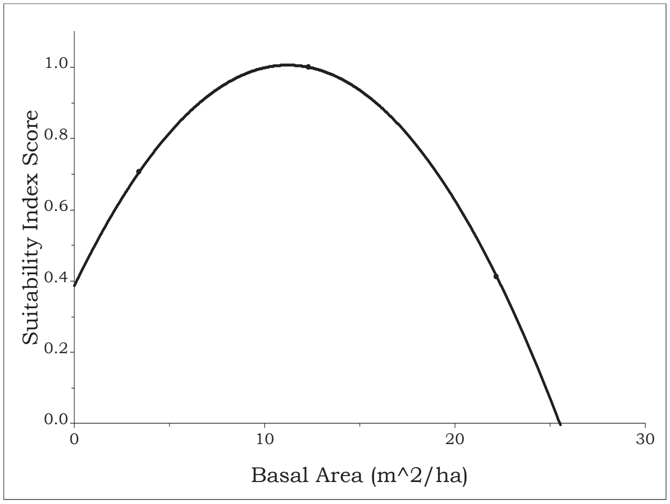


Figure 17.—Relationship between basal area and suitability index (SI) scores for blue-gray gnatcatcher habitat. Equation: SI score = 0.3863 + 0.1105 * (basal area) – 0.0049 * (basal area)$^2$.

**Table 36.—Influence of basal area ($m^2$/ha) on suitability index (SI) scores for blue-gray gnatcatcher habitat**

| Basal area[a] | SI score |
|---|---|
| 3.41 | 0.706 |
| 12.33 | 1.000 |
| 22.20 | 0.412 |

[a] Annand and Thompson (1997).

To calculate the HSI score for sapling, pole, and sawtimber age classes, we determined the geometric mean of SI scores for forest structure (SI1 and SI5) and landscape composition attributes (Max(SI2 or SI3) and SI4) separately and then the geometric mean of these means together. Because edge occurrence (SI4) was not applicable to the shrub-seedling age class, we calculated HSI scores separately for this age class and summed across age classes to determine the overall HSI score for the landscape.

Sapling, pole, and sawtimber successional age classes:

$$HSI_{Old} = (((SI1 * SI5)^{0.500}) * ((Max\ (SI2\ or\ SI3)) * SI4)^{0.500})^{0.500}$$

Shrub-seedling successional age classes:

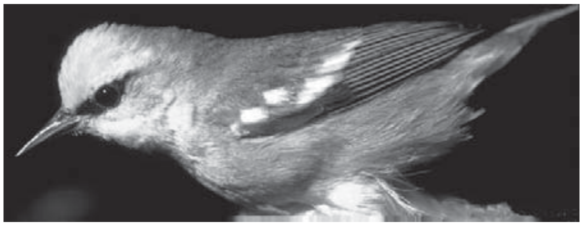
$$HSI_{Shrub} = ((SI1 * SI5)^{0.500} * (Max\ (SI2\ or\ SI3)))^{0.500}$$

$$Overall\ HSI = HSI_{Old} + HSI_{Shrub}$$

## Verification and Validation

The blue-gray gnatcatcher was found in all 88 subsections of the CH and WGCP. Spearman rank correlation analysis on average HSI score and mean BBS route abundance across subsections resulted in a significant ($P \leq 0.001$) positive relationship ($r_s = 0.58$) between these variables. The generalized linear model predicting BBS abundance from BCR and HSI for the blue-gray gnatcatcher was significant ($P \leq 0.001$; $R^2 = 0.210$), and the coefficient on the HSI predictor variable was both positive ($\beta = 19.625$) and significantly different from zero ($P \leq 0.001$). Therefore, we considered the HSI model for the blue-gray gnatcatcher both verified and validated (Tirpak and others 2009a).

## Blue-winged Warbler



Chandler S. Robbins, Patuxent Bird Identification InfoCenter
Photo used with permission

### Status

The blue-winged warbler (*Vermivora pinus*) is a neotropical migrant found from southern New England west to the Lake States and south through the southern Appalachians and Ozarks. Across most of its range, this species has been stable and has even increased in some areas (possibly to the detriment of the golden-winged warbler, with which it sometimes interbreeds; Gill 1980). Once limited to a mostly Midwestern range, this bird expanded into southern New England as forests were cleared and farms were abandoned. However, as the forest has matured in this region, the blue-winged warbler has experienced declines (3.3 and 5.3 percent annually from 1966 to 2004 in the increasingly residential Connecticut and New Jersey, respectively). A similar phenomenon has occurred in the Southeast and BBS data indicate a 3.7 percent decline in FWS Region 4 during this same period (Sauer and others 2005). This species is designated a Bird of Conservation Concern in the CH but not in the WGCP (Table 1), where it rarely breeds. It has a regional combined score of 19 in the CH and requires management attention in that region (Table 1).

### Natural History

The blue-winged warbler is an early successional species (Gill and others 2001) that benefited from European settlement by expanding its range following the initial clearing of forests for agriculture and the subsequent abandonment of farms. Breeding habitat includes early to midsuccessional forest containing dense low growth (shrubs, young trees, thickets). This species makes use of a variety of landform conditions from wetland edges to dry uplands, though mated males have more xeric territories than unmated males. Territories range from 0.2 to 5 ha, with boundaries often aligned along edges. Nests typically are within 30 m of a forest edge in grassy areas with high numbers of small (< 10 cm d.b.h.) trees. Density is inversely related to successional age class, fragmentation, and the abundance of the golden-winged warbler and brown-headed cowbird.

### Model Description

The blue-winged warbler model includes five variables: landform, landcover, successional age class, early successional patch size, and canopy cover.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 37). We directly assigned SI scores to these combinations based on habitat associations reported in Hamel (1992) for the blue-winged warbler. We modified Hamel's data to maximize SI scores in the transitional-shrubland landcover class in the xeric landform.

We also included early successional patch size (SI2) in our model on the basis of data from Rodewald and Vitz (2005) on the relative abundance of the blue-winged warbler in small and large clearcuts (Table 38; Fig. 18). We defined early successional forest by age class and included only grass-forb, shrub-seedling, and sapling age classes in the calculation of patch area.

**Table 37.—Relationship of landform, landcover type, and successional age class to suitability index scores for blue-winged warbler habitat**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.333 | 0.167 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.333 | 0.167 | 0.000 | 0.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.333 | 0.167 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.167 | 0.083 | 0.000 | 0.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.667 | 0.333 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.667 | 0.333 | 0.000 | 0.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.333 | 0.167 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.333 | 0.167 | 0.000 | 0.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.333 | 0.167 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.333 | 0.167 | 0.000 | 0.000 |

**Table 38.—Influence of early successional patch size on suitability index scores for blue-winged warbler habitat; early successional patches include all adjacent grass-forb, shrub-seedling, and sapling successional age class forest**

| Early successional patch size (ha) | SI score |
|---|---|
| 0[a] | 0.000 |
| 4[b] | 0.786 |
| 13[b] | 1.000 |

[a] Assumed value.

[b] Rodewald and Vitz (2005).

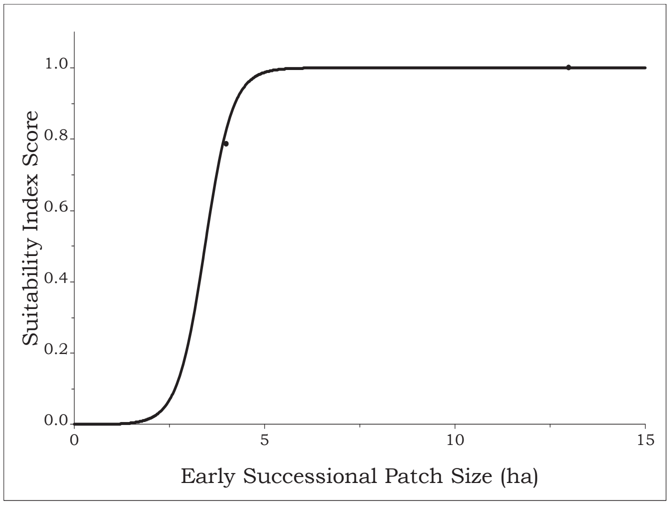


Figure 18.—Relationship between early successional patch size and suitability index (SI) scores for blue-winged warbler habitat. Equation: SI score = 1.000 / (1 + (14353.617 * e $^{-2.788 \text{ * forest patch size}}$)).

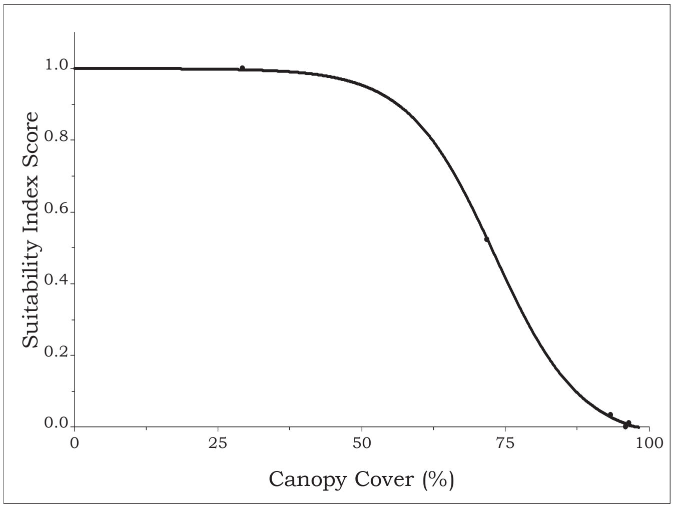


Figure 19.—Relationship between canopy cover and suitability index (SI) scores for blue-winged warbler habitat. Equation: SI score = 1 - (1.0381 / (1 + (16277.383 * $e^{-0.1327 * \text{canopy cover}}$))).

**Table 39.—Influence of canopy cover on suitability index (SI) scores for blue-winged warbler habitat**

| Canopy cover (percent)[a] | SI score |
|---|---|
| 29.26 | 1.000 |
| 71.86 | 0.523 |
| 93.38 | 0.034 |
| 95.58 | 0.000 |
| 96.59 | 0.011 |

[a] Annand and Thompson (1997).

We used an inverse logistic function (Fig. 19) to quantify the relationship between canopy cover (SI3) and SI scores to reflect the lower densities of the blue-winged warbler in forests with increasingly closed canopies. We defined this function by fitting a curve to data from Annand and Thompson (1997) on the relative density of this bird in forest stands with different estimates of canopy cover (Table 39).

To calculate the overall HSI score for this species , we determined the geometric mean of SI scores for forest structure attributes (SI1 and SI3) and then calculated the geometric mean of this value and early successional patch size (SI2).

Overall HSI = $((SI1 * SI3)^{0.500} * SI2)^{0.500}$

## Verification and Validation

The blue-winged warbler was found in 64 of the 88 subsections within the CH and WGCP. We used Spearman rank correlations between average HSI score and mean BBS route abundance at the subsection scale to verify this model. We observed significant positive relationships when analyses included all subsections ($r_s = 0.26$; $P = 0.014$) or only those subsections where this species was detected ($r_s = 0.28$; $P = 0.026$). The generalized linear model predicting BBS abundance from BCR and HSI for the blue-winged warbler was significant ($P \leq 0.001$; $R^2 = 0.232$), and the coefficient on the HSI predictor variable was positive (β = 1.717) but not significantly different from zero ($P = 0.334$). Therefore, we considered the HSI model for the blue-winged warbler verified but not validated (Tirpak and others 2009a).

## Brown Thrasher


Jeffrey A Spendelow, Patuxent Bird Identification InfoCenter
Photo used with permission

### Status

The brown thrasher (*Toxostoma rufum*) is a short-distance migrant found throughout eastern North America. Although populations in the CH and WGCP declined by 1.4 percent per year between 1966 and 2004 (Table 5), this species is not considered a Bird of Conservation Concern in either BCR (Table 1). The brown thrasher has a regional combined score of 13 and 15 in the WGCP and CH, respectively, and is a species warranting management attention in the CH (Table 1).

### Natural History

A ground-foraging passerine, the brown thrasher is associated with edge habitats throughout the eastern United States and Canada (Cavitt and Haas 2000). Breeding habitat includes a variety of vegetation types, but this species reaches its highest densities in shrublands and midsuccessional forests. Grand and Cushman (2003) found that thrashers in Massachusetts were associated predominately with the amount of scrub oak in the landscape. Rumble and Gobeille (2004) found no significant difference in brown thrasher occurrence among seral stages of cottonwood floodplains in South Dakota, though this bird was detected most often in younger forest classes. Savanna restoration efforts increase thrasher abundance by reducing tree density (Davis and others 2000).

Nests are typically low in a tree or shrub but some may be on the ground. Territory size and thrasher density vary according to habitat quality (0.5 to 1.1 ha and 0.1 to 0.4/ha, respectively). The FWS (Cade 1986) developed an HSI model for this species that included three site-specific variables: density of woody stems, canopy cover, and litter cover.

### Model Description

Our brown thrasher model includes six variables: landform, landcover, successional age class, edge occurrence, small stem density (<2.5 cm d.b.h.), and forest composition in a 10-km radius.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 40). We directly assigned SI scores to these combinations on the basis of habitat associations reported by Hamel (1992) for the brown thrasher in the Southeast.

This edge species inhabits thickets and hedgerows in deciduous forests. Because the brown thrasher uses both hard and soft edges, we defined edge (SI2) as the interface between pole age forest and herbaceous or non-forest landcovers (hard edge) and seedling or grass-forb age forest (soft edge). To be suitable, we required pole age forest sites to be adjacent to an edge (Table 41). However, we relaxed this requirement for seedling-shrub and sapling stands, which we considered suitable regardless of edge.

**Table 40.—Relationship of landform, landcover type, and successional age class to suitability index scores for brown thrasher habitat; values in parentheses apply to West Gulf Coastal Plain/Ouachitas**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.500 | 0.333 | 0.083 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.500 | 0.333 | 0.083 | 0.000 |
| | Deciduous | 0.000 | 0.500 | 0.333 | 0.083 | 0.000 |
| | Evergreen | 0.000 | 0.667 | 0.500 | 0.167 | 0.000 |
| | Mixed | 0.000 | 1.000 | 0.667 | 0.167 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.500 | 0.333 | 0.083 | 0.000 |
| | Woody wetlands | 0.000 | 0.667 | 0.417 | 0.083 | 0.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.667 | 0.417 | 0.083 | 0.000 |
| | Transitional-shrubland | 0.000 | 1.000<br>(0.667) | 0.667<br>(0.500) | 0.167 | 0.000 |
| | Deciduous | 0.000 | 0.667 | 0.417 | 0.083 | 0.000 |
| | Evergreen | 0.000 | 0.667 | 0.500 | 0.167 | 0.000 |
| | Mixed | 0.000 | 1.000 | 0.667 | 0.167 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.667 | 0.417 | 0.083 | 0.000 |
| | Woody wetlands | 0.000 | 0.667 | 0.500 | 0.167 | 0.000 |
| Xeric-ridge | Low-density residential | 0.000 | 1.000 | 0.667 | 0.167 | 0.000 |
| | Transitional-shrubland | 0.000 | 1.000<br>(0.334) | 0.667<br>(0.250) | 0.167<br>(0.083) | 0.000 |
| | Deciduous | 0.000 | 1.000 | 0.667 | 0.167 | 0.000 |
| | Evergreen | 0.000 | 0.667<br>(0.334) | 0.500<br>(0.250) | 0.167<br>(0.083) | 0.000 |
| | Mixed | 0.000 | 1.000 | 0.667 | 0.167 | 0.000 |
| | Orchard-vineyard | 0.000 | 1.000 | 0.667 | 0.167 | 0.000 |
| | Woody wetlands | 0.000 | 0.667 | 0.500 | 0.167 | 0.000 |

**Table 41.—Influence of edge on suitability index (SI) scores for brown thrasher habitat**

| 3 × 3 pixel window around forest pixel includes field[a] | SI score |
|---|---|
| Yes[b] | 1.0 |
| No | 0.0 |

[a] Field defined as any shrub-seedling or grass-forb age class pixel, or natural grasslands, pasture-hay, fallow, urban-recreational grasses, emergent herbaceous wetlands, open water, high intensity residential, commercial-industrial-transportation, bare rock-sand-clay, quarries-strip mines-gravel pits, row crops, or small grains. Forest defined as any used pole age class pixel of low-density residential, transitional, shrublands, deciduous, mixed, evergreen, orchard, or woody wetlands.

[b] Seedling-shrub and sapling habitats used regardless of edge.

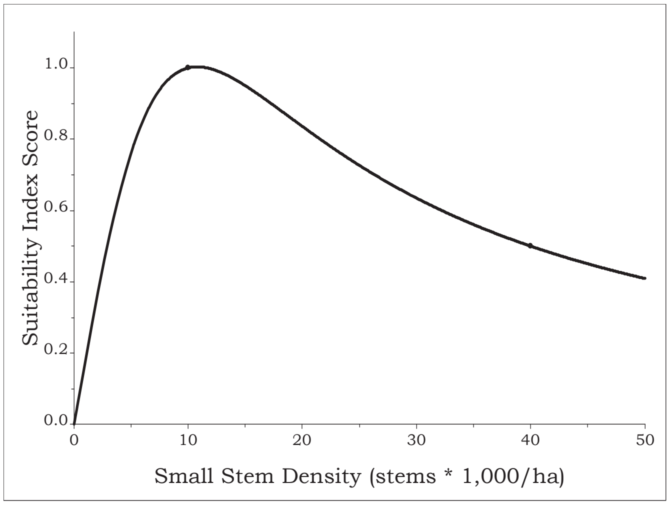


Figure 20.—Relationship between small stem (< 2.5 cm d.b.h.) density (stems * 1000/ha) and suitability index (SI) scores for brown thrasher habitat. Equation: SI score = (0.1 + (0.165 * (small stem density / 1000))) / (1 + (-0.003 * (small stem density / 1000)) + (0.0078 * ((small stem density / 1000))$^2$)).

**Table 42.—Influence of small stem (< 2.5 cm d.b.h.) density (stems * 1,000/ha) on suitability index (SI) scores for brown thrasher habitat**

| Small stem density[a] | SI score |
|---|---|
| 0 | 0.1 |
| 10 | 1.0 |
| 40 | 0.5 |

[a] Cade (1986).

The brown thrasher occupies habitats with numerous small stems (SI3). We fit a smoothed quadratic function (Fig. 20) to HSI cutoff values from the FWS HSI model for this species (Cade 1986; Table 42) to quantify the relationship between small stem density and habitat suitability.

Although the brown thrasher is associated with edges, it prefers modestly forested landscapes (Haas 1997). We included forest composition (SI4) in our model, assuming that habitat suitability would be low if there were no woodland (i.e., 0 percent forest, the left side of the function; Fig. 21) or no edges (i.e., 100 percent forest, the right side of the function). Haas (1997) observed higher reproductive success for birds in more isolated shelterbelts and Robbins and others (1989) observed negative relationships between the occurrence of the gray catbird and American robin (species that share similar habitat preferences to those of the brown thrasher) and forest patch size. Further, Perkins and others (2003b) observed an increase in abundance of edge-associated birds as the total amount of woody cover decreased. However, the brown thrasher responded positively to the amount of forest cover in the study area. We interpreted these observations as evidence that this species would exhibit a preference for landscapes with moderate forest landcover. We fit a Gaussian function to landscape proportions reflecting this pattern and assumed that landscapes that were 70 percent forested were associated with the maximum SI score (Table 43).

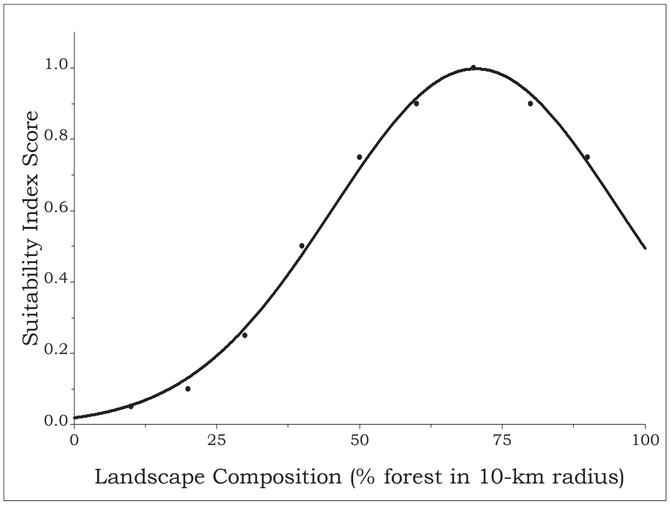


Figure 21.—Relationship between landscape composition and suitability index (SI) scores for brown thrasher habitat. Equation: SI score = $0.998 * e^{((0 - ((\text{landscape composition}) - 70.304) \wedge 2) / 1253.402)}$.

**Table 43.—Relationship between landscape composition (percent forest in 10-km radius) and suitability index (SI) scores for brown thrasher habitat**

| Landscape composition[a] | SI score |
|---|---|
| 0 | 0.00 |
| 10 | 0.05 |
| 20 | 0.10 |
| 30 | 0.25 |
| 40 | 0.50 |
| 50 | 0.75 |
| 60 | 0.90 |
| 70 | 1.00 |
| 80 | 0.90 |
| 90 | 0.75 |
| 100 | 0.50 |

[a] Assumed value.

We assumed that the brown thrasher used edge as a surrogate to early successional habitat, so we calculated HSI scores separately for young (seedling-shrub and sapling) and old (pole) age class forests. In the former, the geometric mean of forest structure and landscape composition variables defines the suitability score. For the latter, we included edge occurrence in the calculation. We summed the age class-specific HSI scores to determine the overall HSI score for all sites.

Seedling-shrub and sapling successional age classes:

$HSI_{Young}$: $((SI1 * SI3)^{0.500} * SI4)^{0.500}$

Pole successional age class:

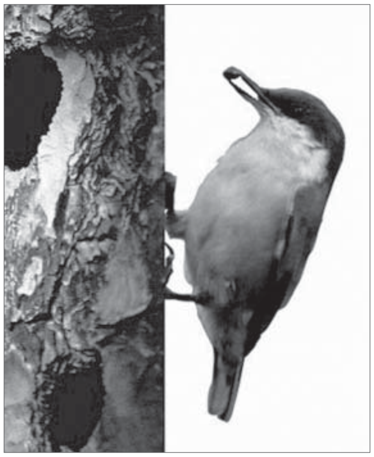
$HSI_{Pole}$: $((SI1 * SI3)^{0.500} * SI4)^{0.500} * SI2$

Overall SI = $HSI_{Young} + HSI_{Pole}$

## Verification and Validation

The brown thrasher was found in all 88 subsections of the CH and WGCP. Spearman rank correlation did not identify a positive relationship between average HSI score and mean BBS route abundance across subsections. The generalized linear model predicting BBS abundance from BCR and HSI for the brown thrasher was significant ($P \leq 0.001$; $R^2 = 0.719$); however, the coefficient on the HSI predictor variable was negative ($\beta = -7.087$). Therefore, we considered the HSI model for the brown thrasher neither verified nor validated (Tirpak and others 2009a).

## Brown-headed Nuthatch


Fernbank Science Center
Photo used with permission

### Status

The brown-headed nuthatch (*Sitta pusilla*) is a resident species of mature pine forests along the Piedmont and Coastal Plains of the southeastern United States. Although this species has experienced modest declines throughout most of its range over the last 40 years (1.2 percent per year), only in Florida has the decline been significant (4.2 percent annually from 1966 to 2004; Sauer and others 2005). This species is an FWS Bird of Conservation Concern in the WGCP (Table 1), where it has a regional combined score of 19. The brown-headed nuthatch is a rare breeder in the CH (regional combined score = 19), and PIF considers this species one that warrants critical recovery in that region.

### Natural History

The brown-headed nuthatch is closely associated with pine: it breeds in mature pine forests and forages almost exclusively in pine trees (> 98 percent of observations; Withgott and Smith 1998). Although often associated with the longleaf pine savanna characteristic of the habitat for red-cockaded woodpecker and Bachman's sparrow, the brown-headed nuthatch has a broader niche than these species (Hamel 1992, Dornak and others 2004). The habitat of this species is defined by two habitat elements: mature pines for foraging and cavities for nesting (Wilson and Watts 1999, Dornak and others 2004). Specific composition of pine species is not as critical as d.b.h., with an average d.b.h. of 25.6 cm considered optimal (O'Halloran and Conner 1987 cited in Dornak and others 2004). The brown-headed nuthatch nests primarily in large-diameter snags < 3 m tall and may require seven to eight snags per ha to ensure adequate nest and roost sites, particularly in the presence of interspecific competition for cavities. In urban areas, the brown-headed nuthatch readily adopts nest boxes and may use other manmade cavities, such as streetlights.

This species prefers open pine stands with few hardwoods (≤ 17.4 stems/ha and basal area ≤ 5 $m^2$/ha) and an open midstory (Wilson and Watts 1999). Optimal canopy cover is highly variable (15 to 85 percent) but stands with closed canopies are not preferred (O'Halloran and Conner 1987, Wilson and Watts 1999). Undergrowth typically is sparse (roughly 35 percent; Dornak and others 2004). The nuthatch regularly breeds at low densities in suboptimal habitats, including stands with small pines, a large fraction of hardwoods, and dense understories (Withgott and Smith 1998). Area sensitivity apparently is not an issue for this species, which is not an acceptable host for the brown-headed cowbird (Withgott and Smith 1998).

### Model Description

The HSI model for the brown-headed nuthatch includes six variables: landform, landcover, successional age class, snag density, small stem (< 2.5 cm d.b.h.) density, and hardwood basal area.

**Table 44.—Relationship of landform, landcover type, and successional age class to suitability index scores for brown-headed nuthatch habitat**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Evergreen | 0.000 | 0.000 | 0.334 | 0.834 | 1.000 |
| | Mixed | 0.000 | 0.000 | 0.167 | 0.333 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Evergreen | 0.000 | 0.000 | 0.334 | 0.834 | 1.000 |
| | Mixed | 0.000 | 0.000 | 0.167 | 0.333 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Evergreen | 0.000 | 0.000 | 0.334 | 0.834 | 1.000 |
| | Mixed | 0.000 | 0.000 | 0.167 | 0.333 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 44). We directly assigned SI scores to these combinations on the basis of habitat associations of the brown-headed nuthatch described by Hamel (1992).

We included snag density (SI2) in our HSI model because of the importance of cavities to this species. We assumed that the SI score was zero when eight or fewer snags of any size were present (Dornak and others 2004). We fit a logistic function (Fig. 22) to data from Wilson and Watts (1999) (Table 45) to quantify the relationship between snag density and SI scores.

We also used small stem density as a function (SI3) in the HSI model to account for the preference of the brown-headed nuthatch for open understories. We fit an inverse logistic function (Fig. 23) to hypothetical data reflecting this preference (Table 46). The shape of this function is supported by observations from Wilson and others (1995), who observed a higher abundance of the brown-headed nuthatch in stands immediately following wildlife stand improvements and prescribed burns (when stem density was lowest) with subsequent declines in abundance as stem density increased through time.

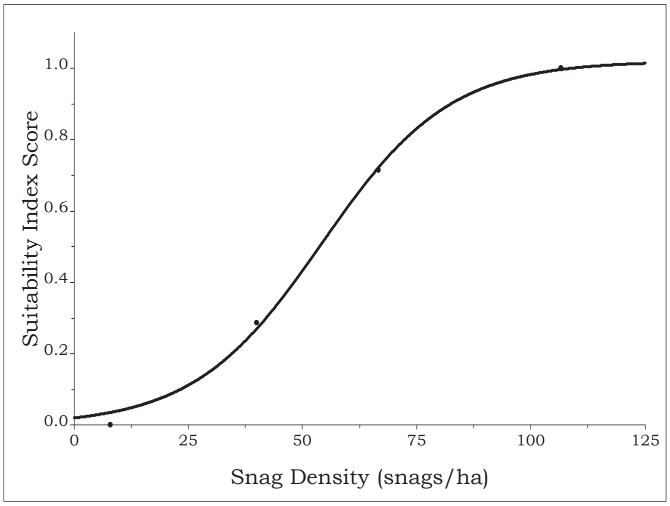


Figure 22.—Relationship between snag density and suitability index (SI) scores for brown-headed nuthatch habitat. Equation: $\text{SI score} = 1.000 / (1 + (49.165 * e^{(-0.073 * \text{snag density})}))$.

**Table 45.—Influence of snag density on suitability index (SI) scores for brown-headed nuthatch habitat**

| Snag density (snags/ha) | SI score |
|---|---|
| 8[a] | 0.000 |
| 40[b] | 0.286 |
| 66.67[b] | 0.715 |
| 106.67[b] | 1.000 |

[a] Dornak and others (2004).
[b] Wilson and Watts (1999).

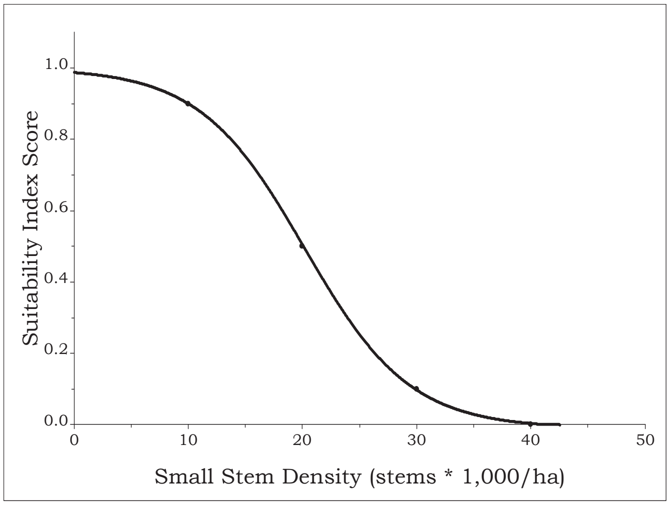


Figure 23.—Relationship between small stem (< 2.5 cm d.b.h.) density (stems * 1000/ha) and suitability index (SI) scores for brown-headed nuthatch habitat. Equation: $\text{SI score} = 1 - (1.010 / (1 + (79.565 * e^{(-0.217 * (\text{small stem density} / 1000))})))$.

**Table 46.—Influence of small stem (< 2.5 cm d.b.h.) density (stems * 1,000/ha) on suitability index (SI) scores for brown-headed nuthatch habitat**

| Small stem density[a] | SI score |
|---|---|
| 0[1] | 1.0 |
| 10[1] | 0.9 |
| 20[1] | 0.5 |
| 30[1] | 0.1 |
| 40[1] | 0.0 |

[a] Assumed value.

Finally, we incorporated hardwood basal area (SI4) as a model variable as birds are less abundant in habitats with a greater hardwood component (Wilson and others 1995, Withgott and Smith 1998, Wilson and Watts 1999). Again, we relied on data from Wilson and Watts (1999) (Table 47) to develop an inverse logistic function to describe the relationship between hardwood basal area and SI score (Fig. 24).

To determine the overall HSI score for the brown-headed nuthatch, we calculated the geometric mean of the four individual functions related to forest structure attributes.

$\text{Overall HSI} = (SI1 * SI2 * SI3 * SI4)^{0.250}$

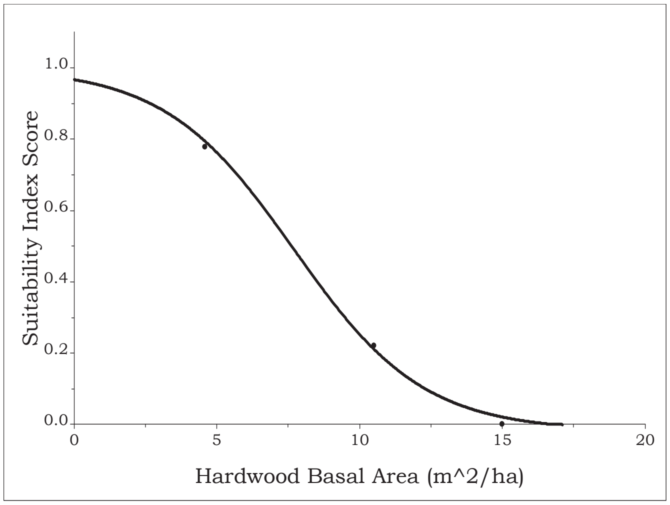


Figure 24.—Relationship between hardwood basal area and suitability index (SI) scores for brown-headed nuthatch habitat. Equation: SI score = 1 - (1.018 / (1 + (29.747 * e $^{(-0.441\ *\ \text{hardwood basal area})}$))).

**Table 47.—Influence of hardwood basal area on suitability index (SI) scores for brown-headed nuthatch habitat**

| Hardwood basal area ($m^2$/ha) | SI score |
|---|---|
| 0.0[a] | 1.000 |
| 4.6[a] | 0.778 |
| 10.5[a] | 0.222 |
| 15.0[b] | 0.000 |
| 20.0[b] | 0.000 |

[a] Wilson and Watts (1999).
[b] Assumed value.

## Verification and Validation

The brown-headed nuthatch was found in 37 of the 88 subsections within the CH and WGCP. Spearman rank correlation identified a significant ($P \leq 0.001$) positive relationship ($r_s = 0.58$) between average HSI score and mean BBS route abundance across subsections. This relationship was even stronger ($r_s = 0.80$) when subsections in which the brown-headed nuthatch was not detected were removed from the analysis. The generalized linear model predicting BBS abundance from BCR and HSI for the brown-headed nuthatch was significant ($P \leq 0.001$; $R^2 = 0.738$), and the coefficient on the HSI predictor variable was both positive ($\beta = 4.712$) and significantly different from zero ($P \leq 0.001$). Therefore, we considered the HSI model for the brown-headed nuthatch both verified and validated (Tirpak and others 2009a).

## Carolina Chickadee

Charles H. Warren, images.nbii.gov

### Status

The Carolina chickadee (*Parus carolinensis*) is a resident species of the southeastern United States. Although populations have been stable in the CH, this species has declined by about 2 percent annually over the last 40 years in the WGCP (Table 5). This bird is a planning and responsibility species in both the CH (regional combined score = 15) and WGCP (regional combined score = 16; Table 1).

### Natural History

The Carolina chickadee is a generalist species that breeds in a variety of forest types across a broad spectrum of landforms (Mostrom and others 2002). It nests in cavities of live and dead trees within multilayered forests containing well developed shrub, midstory, and overstory canopies (Hamel 1992). Abundance declines following reduction of hardwoods in pine stands, likely as a result of the loss of midstory trees (Provencher and others 2002). Nest success and adult survival is positively correlated with woodlot area but is lower on edges regardless of patch size (Doherty and Grubb 2002). Nest destruction by the house wren is a major cause of nest failure in areas where the ranges of these species overlap. Territory size ranges from 1.6 to 2.4 ha.

### Model Description

The Carolina chickadee model includes four variables: landform, landcover, successional age class, and snag density.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 48). We directly assigned SI scores to these combinations on the basis of vegetation and successional age class associations of the Carolina chickadee reported in Hamel (1992).

We included snag density (SI2) as a variable because of the importance of nest and roost cavities for the chickadee, a secondary cavity nester. Data for the Carolina chickadee were not available but Rumble and Gobeille (2004) and Sedgwick and Knopf (1990) observed the black-capped chickadee in habitats with six snags per hectare (Table 49). Therefore, we assumed that stands with six or more snags per ha were representative of optimal habitat. Because the chickadee can use cavities in live trees, we assumed that stands with no snags were not necessarily nonhabitat and assigned to them a small but non-zero SI score (0.03). We fit a logistic function through these data points to quantify the relationship between snag density and habitat suitability (Fig. 25).

We calculated the overall HSI score as the geometric mean of the two individual functions:

Overall HSI = $(SI1 * SI2)^{0.500}$

**Table 48.—Relationship of landform, landcover type, and successional age class to SI scores for Carolina chickadee habitat; values in parentheses apply to West Gulf Coastal Plain/Ouachitas**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| | Deciduous | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| | Evergreen | 0.000 | 0.000 | 0.334 | 0.834 | 1.000 |
| | Mixed | 0.000 | 0.000 | 0.334 | 0.834 | 1.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| | Woody wetlands | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.334 | 0.834 | 1.000 |
| | Deciduous | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| | Evergreen | 0.000 | 0.000 | 0.334 | 0.834 | 1.000 |
| | Mixed | 0.000 | 0.000 | 0.334 | 0.834 | 1.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| | Woody wetlands | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.334<br>(0.250) | 0.834<br>(0.667) | 1.000<br>(0.834) |
| | Deciduous | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| | Evergreen | 0.000 | 0.000 | 0.334<br>(0.250) | 0.834<br>(0.667) | 1.000<br>(0.834) |
| | Mixed | 0.000 | 0.000 | 0.334 | 0.834 | 1.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| | Woody wetlands | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |

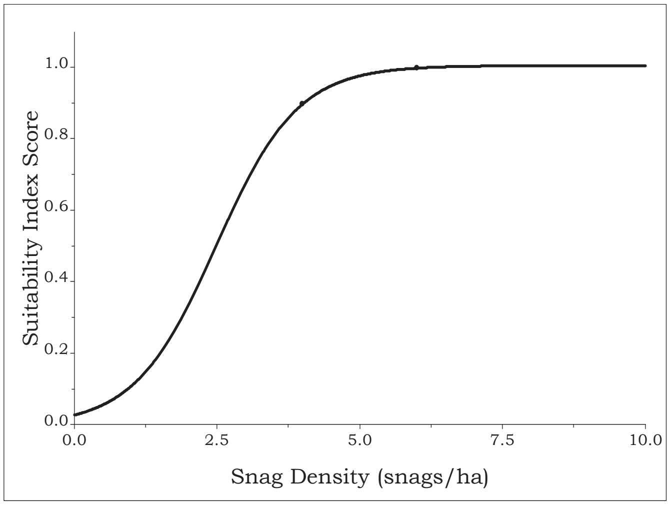


Figure 25.—Relationship between snag density and suitability index (SI) scores for Carolina chickadee habitat. Equation: SI score = 1.007 / (1.000 + (32.567 * e $^{(-1.403 * \text{snag density})}$)).

**Table 49.—Influence of snag density on suitability index (SI) scores for Carolina chickadee habitat**

| Snag density (snags/ha) | SI score |
|---|---|
| 0[a] | 0.03 |
| 4[b] | 0.90 |
| 6[a, c] | 1.00 |

[a] Rumble and Gobeille (2004).

[b] Assumed value.

[c] Sedgwick and Knopf (1990).

## Verification and Validation

The Carolina chickadee was found in all 88 subsections of the CH and WGCP. Spearman rank correlation identified a significant ($P \leq 0.001$) positive relationship ($r_s = 0.55$) between average HSI score and mean BBS route abundance across subsections. The generalized linear model predicting BBS abundance from BCR and HSI for the Carolina chickadee was significant ($P \leq 0.001$; $R^2 = 0.473$), and the coefficient on the HSI predictor variable was both positive ($\beta = 5.142$) and significantly different from zero ($P = 0.038$). Therefore, we considered the HSI model for the Carolina chickadee both verified and validated (Tirpak and others 2009a).

## Cerulean Warbler


U.S. Forest Service

### Status

The cerulean warbler (*Dendroica cerulea*) is a long-distance migrant to the eastern United States. Densities are highest in the Ohio River Valley and along the Cumberland Plateau. This species has declined across most of its range, including the CH and WGCP (6.3 and 9.5 percent per year from 1966 to 2004, respectively; Table 5). The cerulean warbler is classified as a Bird of Conservation Concern requiring critical recovery in the WGCP (regional combined score = 19) and immediate management in the CH (regional combined score = 19) (Table 1). Concern for this species culminated in a petition to the FWS to list the cerulean warbler as threatened. However, this action was deemed unwarranted on the basis of current scientific information (Federal Register 71:234 [6 December 2006] p. 70717).

### Natural History

A forest interior specialist, the cerulean warbler has experienced some of the most dramatic declines of any songbird over the last 30 years (Hamel 2000). This species has a broad geographic range but is abundant only locally. It may nest semi-colonially, with territories in good habitat highly clumped. The cerulean warbler seems to be highly sensitive to forest fragmentation. Robbins and others (1989) found a 50 percent reduction in observations of this species as forest patch size declined from 3,000 to 700 ha. No birds were detected on forest patches less than 138 ha. Estimates from other researchers suggest that forest tracts as large as 8,000 ha may be required to ensure sustainable populations in the Mississippi Alluvial Valley (summarized in Hamel [2000]).

Although it requires large forest tracts, the cerulean warbler establishes territories near interior forest gaps. Weakland and Wood (2005) observed a positive association between this species and forest roads or snags that created small canopy openings. Aside from canopy gaps (a measure of horizontal canopy structure), the cerulean warbler also may respond to the vertical canopy profile. Canopy cover of 6 to 12 m and more than 24 m was preferred in West Virginia (Weakland and Wood 2005). In Ontario, canopy cover of 12 to 18 m and more than 18 m was preferred (Jones and Robertson 2001). The difference in preferred canopy heights between these studies likely reflects differences in local vegetation structure rather than an absolute difference in preferred canopy height. The key habitat feature in both is the multilayered character of the overstory canopy.

Closed-canopy stands with large trees (both in height and d.b.h.) are commonly associated with the cerulean warbler but likely are a crude proxy for the aforementioned canopy features that provide the true selection criteria for this bird (Hamel 2000). This species is associated with bottomland hardwoods in the Southeast and ridges in West Virginia (Hamel 2000, Weakland and Wood 2005). Again, specific landforms probably are not directly selected for but are correlated with the location of large tracts of deciduous forest containing large trees and favorable canopy conditions in these landscapes.

**Table 50.—Relationship of landform, landcover type, and successional age class to suitability index scores for cerulean warbler habitat**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.500 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.400 | 0.800 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.500 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.400 | 0.800 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.400 | 0.800 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.400 | 0.800 |

In “Birds of North America,” Hamel (2000) stated: “Important habitat elements for this species thus appear to be large tracts with big deciduous trees in mature to old-growth forest with horizontal heterogeneity of the canopy. The pattern of vertical distribution of foliage in the canopy is also important.”

## Model Description

The HSI model for the cerulean warbler includes seven variables: landform, landcover, successional age class, forest patch size, percent forest in a 1-km radius, dominant tree density, and canopy cover.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 50). We directly assigned SI scores to these combinations on the basis of habitat associations of the cerulean warbler outlined in Hamel (1992).

We derived the suitability function for forest patch size (SI2) by fitting a logistic curve (Fig. 26) to data from Robbins and others (1989) and Rosenberg and others (2000), who

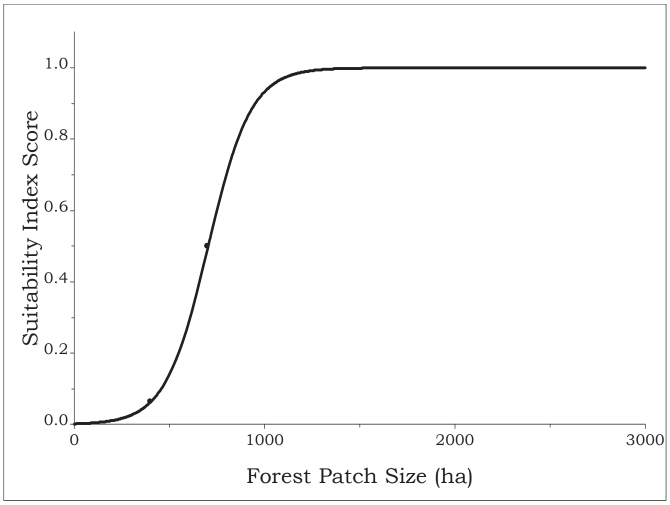


Figure 26.—Relationship between forest patch size and suitability index (SI) scores for cerulean warbler habitat. Equation: SI score = 1.000 / (1.000 + (524.457 * e $^{-0.0089 * \text{forest patch size}}$)).

**Table 51.—Influence of forest patch size on suitability index (SI) scores for cerulean warbler habitat**

| Forest patch size (ha) | SI score |
|---|---|
| 400[a] | 0.064 |
| 700[b] | 0.500 |
| 3,000[b] | 1.000 |
| 5,000[c] | 1.000 |

[a] Rosenberg and others (2000).
[b] Robbins and others (1989).
[c] Assumed value.

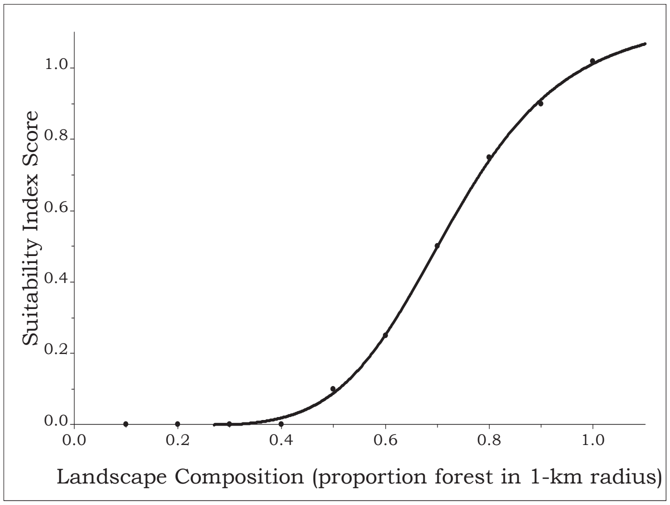


Figure 27.—Relationship between landscape composition and suitability index (SI) scores for cerulean warbler habitat. Equation: SI score = 1.047 / (1.000 + (1991.516 * e $^{-10.673 * \text{landscape composition}}$)).

**Table 52.—Relationship between landscape composition and suitability index (SI) scores for cerulean warbler habitat**

| Landscape composition | SI score |
|---|---|
| 0.00[a] | 0.00 |
| 0.10[a] | 0.00 |
| 0.20[a] | 0.00 |
| 0.30[a] | 0.00 |
| 0.40[a] | 0.00 |
| 0.50[a] | 0.10 |
| 0.60[a] | 0.25 |
| 0.70[b] | 0.50 |
| 0.80[a] | 0.75 |
| 0.90[a] | 0.90 |
| 1.00[a] | 1.00 |

[a] Assumed value.
[b] Donovan and others (1997).

observed that about 95 percent of all birds in FWS Region 4 were on tracts of at least 400 ha (Table 51). Recognizing the suitability of a forest patch is affected by its landscape context (Rosenberg and others 1999), we fit a logistic function (Fig. 27) to data (Table 52) derived from Donovan and others (1997), who observed differences in predator and brood parasite communities among highly fragmented (< 15 percent), moderately fragmented (45 to 50 percent), and lightly fragmented (> 90 percent forest) landscapes. We assumed that the midpoint between moderately and lightly fragmented forest defined the specific cutoff for average (SI score = 0.500) habitat. We used the maximum value from SI2 or SI3 to account for the suitability of small patches in predominantly forested landscapes.

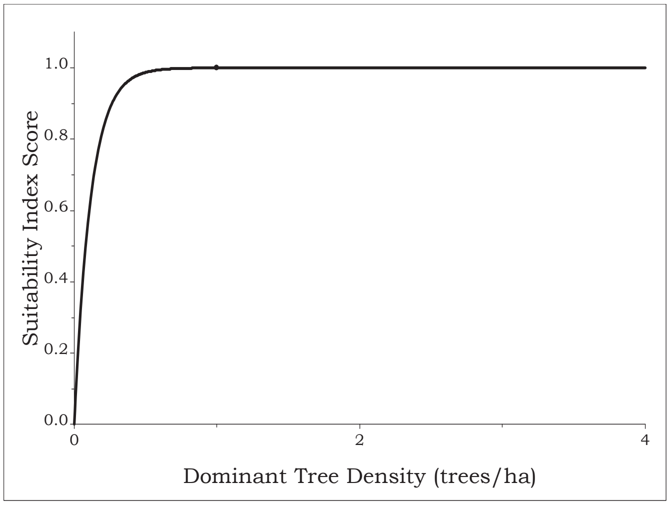


Figure 28.—Relationship between dominant tree density and suitability index (SI) scores for cerulean warbler habitat. Equation: SI score = 1 – e $^{-8.734 * \text{dominant tree density}}$.

**Table 53.—Influence of dominant tree density on suitability index (SI) scores for cerulean warbler habitat**

| Dominant tree density (trees/ha)[a] | SI score |
|---|---|
| 0 | 0.0 |
| 1 | 1.0 |
| 14 | 1.0 |

[a] Assumed value.

We used the density of dominant trees (SI4) in the HSI model and assumed that trees with a d.b.h. greater than 76.2 cm would produce the heterogeneous vertical canopy structure preferred by the cerulean warbler. On the basis of qualitative habitat descriptions by Rosenberg and others (2000), we assumed that the cerulean warbler reached its highest density in stands containing at least one dominant tree per ha. Because this bird nests almost exclusively in these trees (Weakland and Wood 2005), we also assumed that it would be absent from stands with a uniform canopy height (i.e., no dominant trees). We fit an exponential function (Fig. 28) to these data points and assumed that stands with at least 14 dominant trees per ha (the maximum number observed in the WGCP during the FIA surveys of the 1990s) were associated with maximum habitat suitability (Table 53).

We used data from Rosenberg and others (2000), Jones and others (2001), and Weakland and Wood (2005) to derive an inverse quadratic function (Fig. 29) that predicted habitat suitability for the cerulean warbler from canopy cover (SI5; Table 54). Canopy cover of 50 percent or less is associated with failed reproduction by this species (Jones and others 2001), so we considered these values as nonhabitat (SI score = 0.000). Rosenberg and others (2000) identified “a tall, but broken, canopy” as one of the few common denominators of cerulean warbler habitat rangewide, and we maximized the SI score at 90 percent canopy closure. However, Weakland and Wood (2005) observed the cerulean warbler selecting internal edges, so we also discounted habitat suitability for closed canopies. Nonetheless, we recognize that a dense upper canopy is needed by this species (Hamel 2000) and assigned to sites with 80 and 100 percent canopy cover an average SI score (0.500).

To calculate overall HSI scores for cerulean warbler habitat, we calculated the geometric mean of the three suitability indices related to forest structure (SI1, SI4, and SI5) and the maximum value for the two suitability indices related to landscape composition (SI2 and SI3) separately and then the geometric mean of these values together.

$$\text{Overall SI} = ((\text{SI1} * \text{SI4} * \text{SI5})^{0.333} * \text{Max(SI2 or SI3)})^{0.500}$$

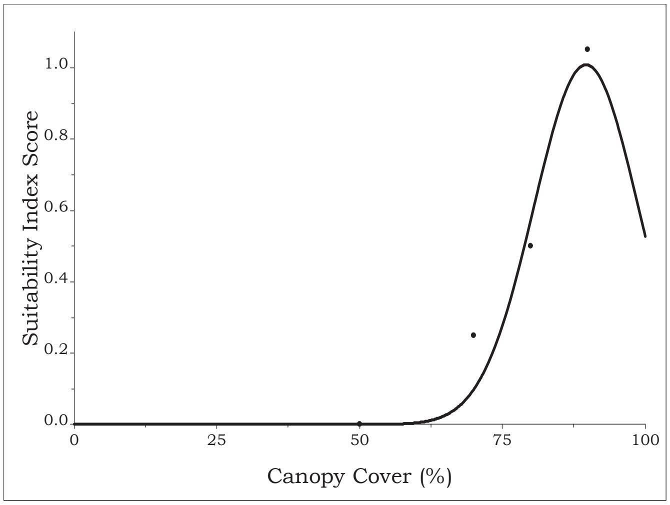


Figure 29.—Relationship between canopy cover and suitability index (SI) scores for cerulean warbler habitat. Equation: SI score = 1 / (62.548 – (1.369 * canopy cover) + (0.007612 * (canopy cover)$^2$)).

**Table 54.—Influence of canopy cover on suitability index (SI) scores for cerulean warbler habitat**

| Canopy cover (percent) | SI score |
|---|---|
| 50[a] | 0.00 |
| 70[b] | 0.25 |
| 80[b] | 0.50 |
| 90[c] | 1.00 |
| 100[d] | 0.50 |

[a] Jones and others (2001).
[b] Hamel (2000).
[c] Rosenberg and others (2000).
[d] Weakland and Wood (2005).

## Verification and Validation

The cerulean warbler was found in 60 of the 88 subsections within the CH and WGCP. Spearman rank correlation identified a significant positive relationship between average HSI score and mean BBS route abundance across all subsections ($P \leq 0.001$; $r_s = 0.44$) and those in which this species was detected ($P \leq 0.001$; $r_s = 0.42$). The generalized linear model predicting BBS abundance from BCR and HSI for the cerulean warbler was significant ($P \leq 0.001$; $R^2 = 0.205$), and the coefficient on the HSI predictor variable was both positive ($\beta = 0.627$) and significantly different from zero ($P = 0.023$). Therefore, we considered the HSI model for the cerulean warbler both verified and validated (Tirpak and others 2009a).

## Chimney Swift

Ron Austing, used with permission

### Status

The chimney swift (*Chaetura pelagica*) is a familiar bird found across most of North America east of the Rocky Mountains. Populations have declined in both the CH and WGCP over the last 40 years (2.6 and 1.1 percent per year). However, the high annual variability in abundance for this species prevents the identification of significant trends (Sauer and others 2005; Table 5). This bird has a regional combined score of 16 and requires management attention in the CH. However, in the WGCP, the chimney swift is only a planning and responsibility species with a regional combined score of 14 (Table 1).

### Natural History

The range of the chimney swift, a small, long-distance migrant, expanded dramatically with European settlement and the increase in artificial nest structures (e.g., chimneys) that followed (Cink and Collins 2002). Prior to European settlement, this species probably was distributed thinly and relied on tree cavities for nesting. Nesting in trees is now rare (Graves 2004) and most nests and roosts are concentrated in urban areas (Cink and Collins 2002). This species is weakly territorial (typically one nest per cavity), and population declines may be due to the loss of nest sites as large, open chimneys become scarce. Home ranges are largely unknown.

### Model Description

For a bird that occurs in such close association with humans, few data are available on the habitat preferences of the chimney swift. We assumed that habitat suitability for this species was primarily a function of the availability of nest and roost sites within the proper landscape context (i.e., open chimneys near foraging areas). To identify these locations, we estimated the proportion of foraging habitats in a 1-km buffer around each pixel of developed landcover. We assumed that this bird could travel 1 km from nesting-roosting areas to foraging habitats (defined as water, grassland, pasture-hay, recreational grasses, or forest landcover classes) and that these habitats had to be more than 1 ha to accommodate the aerial foraging maneuvers of this species. Because the chimney swift is semi-colonial, we also assumed that that as foraging habitat increased in the 1-km buffer, developed pixels were increasingly isolated and would be of lower suitability (Table 55). We used a quadratic curve (Fig. 30) to quantify the relationship between landscape composition and habitat suitability for this species.

### Verification and Validation

The chimney swift occurred in all 88 subsections of the CH and WGCP. Spearman rank correlation identified a significant ($P \leq 0.001$) positive relationship ($r_s = 0.50$) between average HSI score and mean BBS route abundance across subsections. The generalized linear model predicting BBS abundance from BCR and HSI for the chimney swift was significant ($P \leq 0.001$; $R^2 = 0.208$), and the coefficient on the HSI predictor variable was positive ($\beta = 5.043$) but not significantly different from zero ($P = 0.524$). Therefore, we considered the HSI model for the chimney swift verified but not validated (Tirpak and others 2009a).

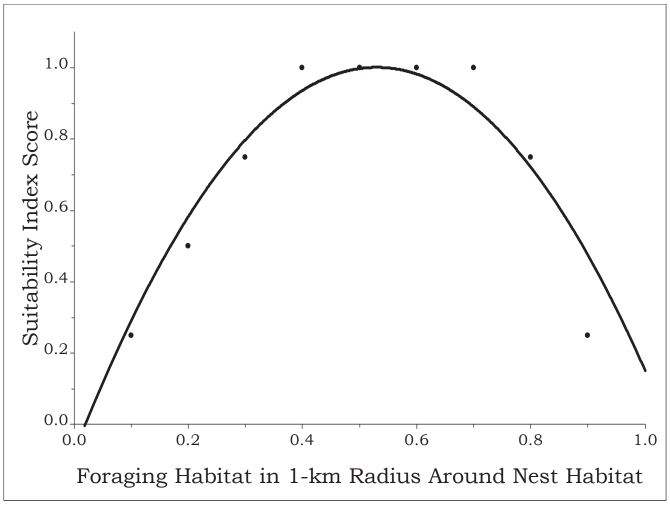


Figure 30.—Relationship between proportion of foraging habitat within 1-km buffer around potential nesting/roosting sites on suitability index (SI) scores for chimney swift habitat. Equation: SI score = (-0.0769 + (4.0734 * proportion foraging cover) - (3.8462 * (proportion foraging cover$^2$))).

**Table 55.—Influence of proportion of foraging habitat[a] within 1-km buffer around potential nesting-roosting sites[b] on suitability index (SI) scores for chimney swift habitat**

| Proportion[c] of foraging habitat around potential nesting-roosting sites | SI score |
|---|---|
| 0.0 | 0.00 |
| 0.1 | 0.25 |
| 0.2 | 0.50 |
| 0.3 | 0.75 |
| 0.4 | 1.00 |
| 0.5 | 1.00 |
| 0.6 | 1.00 |
| 0.7 | 1.00 |
| 0.8 | 0.75 |
| 0.9 | 0.25 |
| 1.0 | 0.25 |

[a] Foraging habitat = water, grassland, pasture-hay, recreational grasses, forest > 1 ha.

[b] Nesting-roosting site = any developed landcover.

[c] Assumed value.

## Chuck-will's-widow


Chandler S. Robbins, Patuxent Bird Identification InfoCenter
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### Status

The chuck-will's-widow (*Caprimulgus carolinensis*) is a neotropical migrant that breeds in the southeastern United States. It has experienced small yet significant declines in the WGCP over the last 40 years (1.3 percent per year; Sauer and others 2005). Populations in the CH have remained relatively stable during the same period (Table 5). Chuck-will's-widow is as Bird of Conservation Concern and a PIF species in need of management attention in the WGCP (regional combined score = 16). This species has no special conservation status in the CH (regional combined score = 14; Table 1).

### Natural History

The chuck-will's-widow, like all nightjars, is nocturnal and most active on moonlit nights. Because of this behavior and its cryptic coloration, this species is difficult to study and few systematic investigations of its habitat, demography, or population status have been conducted. Most of the information on chuck-will's-widow is anecdotal and coincident to studies of other species (Straight and Cooper 2000).

The chuck-will's-widow occupies woodland habitats interspersed with large openings in which the bird forages at night. Calling males are equally abundant among suburban, pasture, and forested landscapes (Cooper 1981). Urban habitats are unsuitable (Straight and Cooper 2000). The chuck-will's-widow prefers more open habitats than the whip-poor-will (Cooper 1981) and is unaffected by forest fragmentation (it may even benefit from it). Drier sites also are preferred.

### Model Description

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 56). We directly assigned SI scores to these combinations on the basis of data from Hamel (1992) on the habitat associations of the chuck-will's-widow in the Southeast.

The realized suitability of the sites identified in SI1 depends largely on landscape context. Cooper (1981) found that the abundance of chuck-will's-widow was highest in areas with equal amounts of forest and agriculture. Therefore, we used the proportion of these two habitats in a 500-m radius window (SI2) in the HSI model. We assigned the maximum SI score to landscapes characterized by 50 percent forest and 50 percent agriculture. We reduced these scores as landscapes varied from this optimal configuration towards a more open or a more forested composition with a stronger reduction in suitability for increasingly forested landscapes (Table 57).

The overall HSI score for chuck-will's-widow is based solely on SI2, which incorporates the results from SI1.

Overall HSI = SI2

**Table 56.—Relationship of landform, landcover type, and successional age class to suitability index scores for chuck-will's-widow habitat; values in parentheses apply to West Gulf Coastal Plain/Ouachitas**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.334 | 0.834 | 1.000 |
| | Deciduous | 0.000 | 0.000 | 0.083 | 0.167 | 0.167 |
| | Evergreen | 0.000 | 0.000 | 0.334 | 0.834 | 1.000 |
| | Mixed | 0.000 | 0.000 | 0.334 | 0.834 | 1.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.334 | 0.834 | 1.000 |
| | Deciduous | 0.000 | 0.000 | 0.083 | 0.167 | 0.167 |
| | Evergreen | 0.000 | 0.000 | 0.334 | 0.834 | 1.000 |
| | Mixed | 0.000 | 0.000 | 0.334 | 0.834 | 1.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.334<br>(0.250) | 0.834<br>(0.583) | 1.000<br>(0.667) |
| | Deciduous | 0.000 | 0.000 | 0.167 | 0.333 | 0.333 |
| | Evergreen | 0.000 | 0.000 | 0.334<br>(0.250) | 0.834<br>(0.583) | 1.000<br>(0.667) |
| | Mixed | 0.000 | 0.000 | 0.334 | 0.834 | 1.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |

## Verification and Validation

The chuck-will's-widow was found in 86 of the 88 subsections within the CH and WGCP. Spearman rank correlations yielded similar results when analysis included all subsections and only those subsections in which this species was detected: significant ($P \leq 0.001$ and 0.003, respectively) positive associations ($r_s = 0.34$ and 0.32, respectively) between average HSI score and mean BBS route abundance. The generalized linear model predicting BBS abundance from BCR and HSI for the chuck-will's-widow was significant ($P \leq 0.001$; $R^2 = 0.312$), and the coefficient on the HSI predictor variable was positive ($\beta = 0.569$) but not significantly different from zero ($P = 0.415$). Therefore, we considered the HSI model for the chuck-will's-widow verified but not validated (Tirpak and others 2009a).

**Table 57.—Suitability index scores for chuck-will's-widow habitat based on proportion of nesting-roosting and foraging habitat within 500-m radius landscape**

| | Proportion foraging[a] | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Proportion nest and roost[b] | 0.0 | 0.1 | 0.2 | 0.3 | 0.4 | 0.5 | 0.6 | 0.7 | 0.8 | 0.9 | 1.0 |
| 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| 0.1 | 0.0 | 0.0 | 0.0 | 0.1 | 0.2 | 0.2 | 0.2 | 0.2 | 0.2 | 0.2 | |
| 0.2 | 0.0 | 0.0 | 0.1 | 0.2 | 0.4 | 0.6 | 0.6 | 0.6 | 0.5 | | |
| 0.3 | 0.0 | 0.1 | 0.2 | 0.4 | 0.6 | 0.6 | 0.8 | 0.8 | | | |
| 0.4 | 0.0 | 0.2 | 0.4 | 0.6 | 0.8 | 0.8 | 1.0 | | | | |
| 0.5 | 0.0 | 0.2 | 0.4 | 0.6 | 0.8 | 1.0[c] | | | | | |
| 0.6 | 0.0 | 0.2 | 0.4 | 0.6 | 0.8 | | | | | | |
| 0.7 | 0.0 | 0.2 | 0.4 | 0.6 | | | | | | | |
| 0.8 | 0.0 | 0.2 | 0.4 | | | | | | | | |
| 0.9 | 0.0 | 0.2 | | | | | | | | | |
| 1.0 | 0.0 | | | | | | | | | | |

[a] Foraging = pasture-hay, recreational grasses, grasslands, and emergent herbaceous wetland landcovers or grass-forb and shrub-seedling successional age classes.

[b] Nest and roost = habitats identified in SI1 (Table 56).

[c] Cooper (1981).

# Eastern Wood-pewee

Jeffrey A Spendelow, Patuxent Bird Identification InfoCenter
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## Status

The eastern wood-pewee (*Contopus virens*) is a long-distance neotropical migrant that breeds throughout the temperate regions of eastern North America (McCarty 1996). This species reaches its highest densities in the Ozark Mountain region of the CH, where it has a regional combined score of 15 (Table 1). In the WGCP, the eastern wood-pewee has a regional combined score of 16. This bird is one requiring management attention in both BCRs, with declining populations in both regions (Sauer and others 2005) (Table 5).

## Natural History

The eastern wood-pewee is a common species in woodlands of all types (deciduous, mixed, and evergreen). However, this species consistently selects open park-like conditions on xeric sites with limited canopy cover and low shrub densities (Robbins and others 1989; McCarty 1996). The eastern wood-pewee is positively associated with increasing density of sawtimber trees, reaching a threshold at 100 trees per ha where a negative relationship develops (Best and Stauffer 1986, Robbins and others 1989).

The eastern wood-pewee, common in both forest interiors and edges, generally is area-insensitive, and may occupy fragments as small as 0.3 ha (Blake and Karr 1987, Robbins and others 1989). Its cryptic nests high in the canopy may limit predation and parasitism, allowing the pewee to occupy small fragments without the adverse effects on reproduction common to other open-cup nesters (McCarty 1996, Knutson and others 2004, Underwood and others 2004). This species is not found in riparian corridors with less than 24 percent forest cover in the landscape (Perkins and others 2003b).

## Model Description

The HSI model for the eastern wood-pewee includes five variables: landform, landcover, successional age class, percent forest in a 1-km radius, and density of sawtimber trees (> 28 cm d.b.h.).

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 58). We directly assigned SI scores to these combinations on the basis of habitat associations of the eastern wood-pewee reported by Hamel (1992).

This species can occupy small forest fragments but may require a minimum amount of forest in the landscape. Therefore, our model did not include a forest patch size function but relied solely on landscape composition (SI2). We used a logistic function (Fig. 31) to predict SI scores from the percentage of forest in the landscape (Table 59).

**Table 58.—Relationship of landform, landcover type, and successional age class to suitability index scores for eastern wood-pewee habitat. Values in parentheses apply to West Gulf Coastal Plain/Ouachitas**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.167 | 0.250 | 0.500 | 0.667 |
| | Transitional-shrubland | 0.000 | 0.167 | 0.250 | 0.500 | 0.667 |
| | Deciduous | 0.000 | 0.167 | 0.250 | 0.500 | 0.667 |
| | Evergreen | 0.000 | 0.250 | 0.333 | 0.667 | 1.000 |
| | Mixed | 0.000 | 0.000 | 0.167 | 0.667 | 1.000 |
| | Orchard-vineyard | 0.000 | 0.167 | 0.250 | 0.500 | 0.667 |
| | Woody wetlands | 0.000 | 0.250 | 0.333 | 0.417 | 0.500 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.167 | 0.583 | 0.834 |
| | Transitional-shrubland | 0.000 | 0.000<br>(0.333) | 0.167<br>(0.333) | 0.667 | 1.000 |
| | Deciduous | 0.000 | 0.000 | 0.167 | 0.583 | 0.834 |
| | Evergreen | 0.000 | 0.250 | 0.333 | 0.667 | 1.000 |
| | Mixed | 0.000 | 0.000 | 0.167 | 0.667 | 1.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.167 | 0.583 | 0.834 |
| | Woody wetlands | 0.000 | 0.250 | 0.333 | 0.500 | 0.667 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.167 | 0.667 | 1.000 |
| | Transitional-shrubland | 0.000 | 0.000<br>(0.167) | 0.167<br>(0.250) | 0.667 | 1.000 |
| | Deciduous | 0.000 | 0.000 | 0.167 | 0.667 | 1.000 |
| | Evergreen | 0.000 | 0.250<br>(0.167) | 0.333<br>(0.250) | 0.667 | 1.000 |
| | Mixed | 0.000 | 0.000 | 0.167 | 0.667 | 1.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.167 | 0.667 | 1.000 |
| | Woody wetlands | 0.000 | 0.250 | 0.333 | 0.500 | 0.667 |

We included density of sawtimber trees in the HSI model and used the threshold of 100 trees per ha observed by Best and Stauffer (1986) as the optimal value in a quadratic function (Fig. 32) that links density of sawtimber trees (SI3) to habitat suitability. Because Best and Stauffer (1986) observed a reduction in wood-pewee abundance at sawtimber tree densities less than 100 trees per ha and Robbins and others (1989) observed a negative relationship between occurrence and tree density, we assumed a symmetrical decline in habitat quality as sawtimber tree density increased or decreased above or below the optimum (Table 60).

To calculate the overall HSI score, we determined the geometric mean of individual SI functions relating to forest structure (SI1 and SI3) and then calculated the geometric mean of this value and landscape composition (SI2).

$$\text{Overall HSI} = ((\text{SI1} * \text{SI3})^{0.500} * \text{SI2})^{0.500}$$

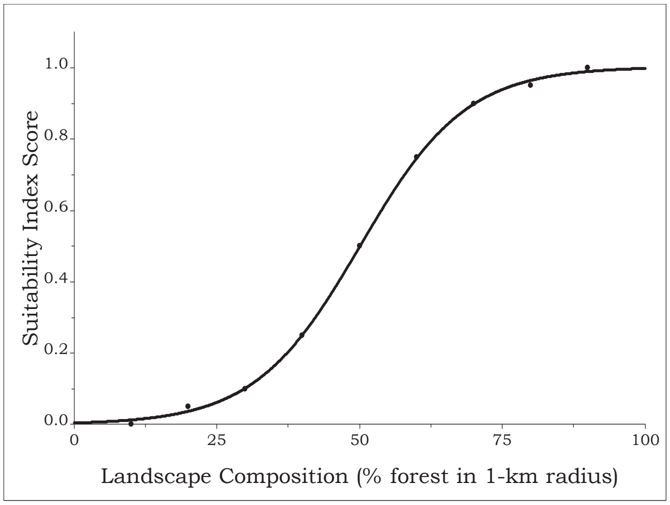

Figure 31.—Relationship between landscape composition and suitability index (SI) scores for eastern wood-pewee habitat. Equation: SI score = $1.005 / (1.000 + (221.816 * e^{-0.108 * (\text{landscape composition})}))$.

**Table 59.—Relationship between landscape composition (percent forest in 1-km radius) and suitability index (SI) scores for eastern wood-pewee habitat**

| Landscape composition | SI score |
|---|---|
| 0[a] | 0.00 |
| 10[a] | 0.00 |
| 20[a] | 0.05 |
| 30[b] | 0.10 |
| 40[a] | 0.25 |
| 50[b] | 0.50 |
| 60[a] | 0.75 |
| 70[b] | 0.90 |
| 80[a] | 0.95 |
| 90[b] | 1.00 |
| 100[a] | 1.00 |

[a] Assumed value.

[b] Dononvan and others (1997).

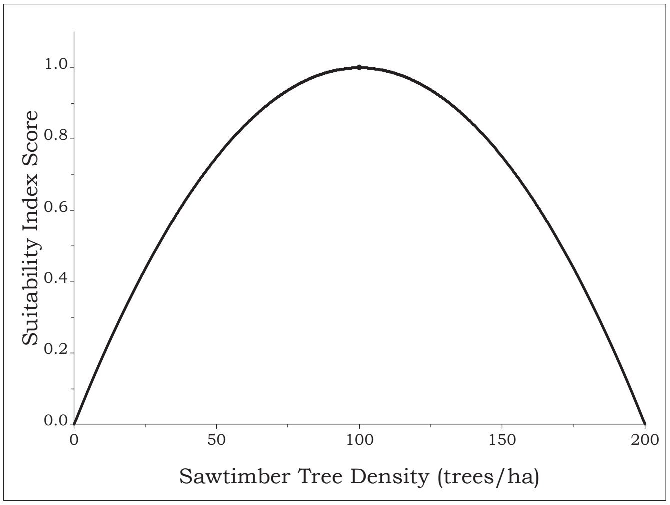

Figure 32.—Relationship between sawtimber tree (≥ 28 cm d.b.h.) density and suitability index (SI) scores for eastern wood-pewee habitat. Equation: SI score = $(0.0200 * \text{sawtimber tree density}) - (0.0001 * (\text{sawtimber tree density}^2))$.

**Table 60.—Influence of sawtimber tree (≥ 28 cm d.b.h.) density (trees/ha) on suitability index (SI) scores for eastern wood-pewee habitat**

| Sawtimber tree density | SI score |
|---|---|
| 0[a] | 0.0 |
| 100[b] | 1.0 |
| 200[a] | 0.0 |

[a] Assumed value.

[b] Best and Stauffer (1986).

## Verification and Validation

The eastern wood-pewee was found in all 88 subsections of the CH and WGCP. Spearman rank correlation on average HSI score and mean BBS route abundance identified a significant ($P \leq 0.001$) positive association ($r_s = 0.46$) between these two variables at the subsection scale. The generalized linear model predicting BBS abundance from BCR and HSI for the eastern wood-pewee was significant ($P \leq 0.001$; $R^2 = 0.472$), and the coefficient on the HSI predictor variable was both positive ($\beta = 5.183$) and significantly different from zero ($P \leq 0.001$). Therefore, we considered the HSI model for the eastern wood-pewee both verified and validated (Tirpak and others 2009a).

# Field Sparrow

Deanna K. Dawson, Patuxent Bird Identification InfoCenter
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## Status

The field sparrow (*Spizella pusilla*) is a short-distance migrant found throughout North America east of the Rocky Mountains. Associated with early successional habitats, this species has experienced the sharp declines typical of many scrub-shrub and grassland species in the East. BBS data indicate declines in populations of the field sparrow in both the CH and WGCP (Sauer and others 2005; Table 5). The field sparrow has a regional combined score of 17 and 15 in the CH and WGCP, respectively, but is not a Bird of Conservation Concern in either BCR (Table 1). About 20 percent of the continental population occurs in the CH (Panjabi and others 2001).

## Natural History

The field sparrow breeds in a variety of vegetation types, including brushy pastures, second-growth scrub, forest openings and edges, Christmas tree farms, orchards, nurseries, and roadsides and railroads near open fields (Carey and others 1994). Abundance increases in forested landscapes managed for early successional habitat (Yahner 2003), and this bird commonly occupies reclaimed mines (DeVault and others 2002) and savanna restoration sites (Davis and others 2000). Abundance is positively related to the size of old fields in Arkansas (Bay 1994). The field sparrow nests on or near the ground in early spring but may nest in saplings or shrubs later in the year. Brood parasitism rates vary geographically but the field sparrow generally is a poor cowbird host. Parasitism rates are higher in thinned forest stands than in regenerating plantations (Barber and others 2001).

This species also uses grasslands, though at lower densities than in shrub-scrub habitats (Horn and others 2002). Grass type affects habitat suitability, with warm-season grasses supporting higher abundance (Giuliano and Daves 2002, Walk and Warner 2000), nest density (Farrand 2005), and productivity than cool-season grasses (Giuliano and Daves 2002). Conservation Reserve Program fields serve as source habitat for the field sparrow in Missouri (McCoy and others 1999).

## Model Description

The model predicting habitat suitability for the field sparrow includes six variables: landform, land cover, successional age class, canopy cover, density of small stems (< 2.5 cm d.b.h.), and the presence of grassy landcover.

The first suitability function of the field sparrow HSI model combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 61). We used habitat associations of the field sparrow reported by Hamel (1992) to assign SI scores to these combinations.

**Table 61.—Relationship of landform, landcover type, and successional age class to suitability index scores for field sparrow habitat**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.333 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.333 | 0.000 | 0.000 | 0.000 |
| | Evergreen | 0.667 | 1.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.667 | 1.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.333 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.167 | 0.000 | 0.000 | 0.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.667 | 1.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.667 | 0.000 | 0.000 | 0.000 |
| | Evergreen | 0.667 | 1.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.667 | 1.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.667 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.333 | 0.000 | 0.000 | 0.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.667 | 1.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 1.000 | 0.000 | 0.000 | 0.000 |
| | Evergreen | 0.667 | 1.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.667 | 1.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 1.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.333 | 0.000 | 0.000 | 0.000 |

We included canopy cover (SI2) and small stem density (SI3) as SIs in our model to account for the absence of the field sparrow from closed-canopy forests or forested sites with an open understory. We used data from Annand and Thompson (1997) (Tables 62 and 63) to fit a quadratic function to canopy cover and a Gaussian function to small stem density for predicting SI scores (Fig. 33 and 34). The negative relationship between the field sparrow and stem density is supported by Carey and others (1994), who observed a reduction in habitat suitability as "thickets of trees spread in the habitat." Sousa (1983) constructed an HSI model that contained a negative relationship between habitat suitability and percent shrub cover. Suitability of habitat for the field sparrow declined from optimal at 50 percent shrub cover (defined as the percentage of ground shaded by a vertical projection of the canopies of woody vegetation less than 5 m) to unsuitable at 75 percent shrub cover. We did not have a quantitative estimate of the relationship between small stem density and shrub cover, so we assumed that 40,000 stems per ha would shade 75 percent of the ground. We were conservative with this estimate; lacking quantitative data, we did not want to exclude stands that might provide habitat for this species.

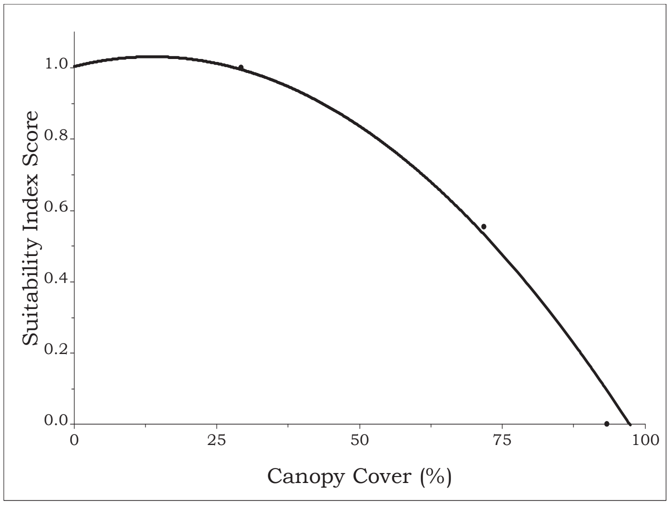

Figure 33.—Relationship between canopy cover and suitability index (SI) scores for field sparrow habitat. Equation: SI score = 1.0038 + 0.0040 * (canopy cover) – 0.0001475 * (canopy cover)$^2$.

**Table 62.—Influence of canopy cover on suitability index (SI) scores for field sparrow habitat**

| Canopy cover (percent) | SI score |
|---|---|
| 0.00[a] | 1.000 |
| 29.26[b] | 1.000 |
| 71.86[b] | 0.555 |
| 93.38[b] | 0.000 |
| 100.00[a] | 0.000 |

[1] Assumed value.
[2] Annand and Thompson (1997).

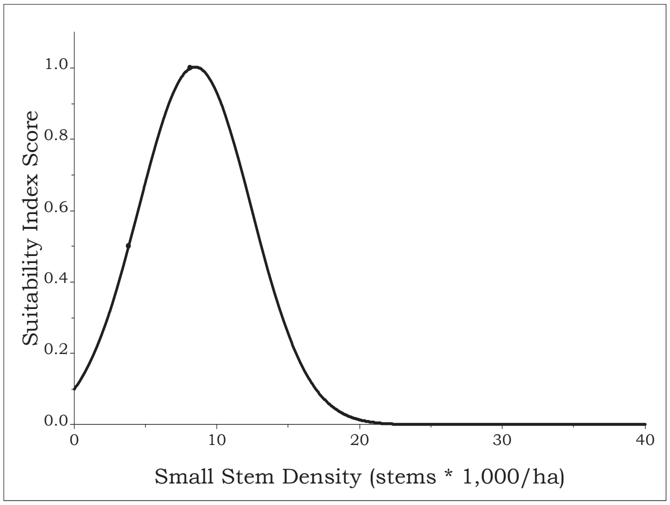

Figure 34.—Relationship between small stem (< 2.5 cm d.b.h.) density (stems * 1000/ha) and suitability index (SI) scores for field sparrow habitat. Equation: SI score = 1.003 * e $(^{-((\text{small stem density} / 1000) – 8.461)\text{^}2 )/ 31.0472}$.

**Table 63.—Influence of small stem (< 2.5 cm d.b.h.) density (stems * 1,000/ha) on suitability index (SI) scores for field sparrow habitat**

| Small stem density | SI score |
|---|---|
| 0[a] | 0.1 |
| 3.812[b] | 0.5 |
| 8.148[b] | 1.0 |
| 40.000[a] | 0.0 |

[a] Sousa (1983).
[b] Annand and Thompson (1997).

**Table 64.—Relationship between grass landcover and suitability index (SI) scores for field sparrow habitat**

| Landcover | SI score |
|---|---|
| Grassland-herbaceous[a] | 1.0 |
| Pasture-hay[a] | 0.5 |

[a] Must occur ≤ 170 meters from forested landcover.

The field sparrow often is associated with grasslands with sufficient perches (Carey and others 1994, Kahl and others 1985). Therefore, we included an SI function related to grasslands (SI4) in the model. Many useable grassland sites may have insufficient woody cover to be classified as shrublands in the NLCD, so we required all grassland types (natural as well as pasture and hayfields) to be within 170 m of a wooded edge—a distance approximating a large field sparrow territory (Best 1974)—to be considered useable. Natural grasslands also are more likely to contain dense grass nesting sites than pastures and hayfields (Giuliano and Daves 2002, Farrand 2005), so we assigned to useable natural grasslands an SI score of 1.000 and to useable pasture-hayfields a score of 0.500 (Table 64).

To calculate the HSI score for field sparrow habitat in forested landcovers, we calculated the geometric mean of the SI scores relating to forest structure (SI1, SI2, and SI3). We added the SI score for grasslands (SI4) to this value to determine the overall HSI score.

Overall HSI = $((SI1 * SI2 * SI3)^{0.333} + SI4)$

## Verification and Validation

The field sparrow was found in 87 of the 88 subsections within the CH and WGCP. Spearman rank correlation on average HSI score and mean BBS route abundance identified a significant ($P \leq 0.001$) positive association ($r_s = 0.55$) between these two variables within subsections where this species was detected. The generalized linear model predicting BBS abundance from BCR and HSI for the field sparrow was significant ($P \leq 0.001$; $R^2 = 0.690$), and the coefficient on the HSI predictor variable was both positive ($\beta = 37.060$) and significantly different from zero ($P \leq 0.001$). Therefore, we considered the HSI model for the field sparrow both verified and validated (Tirpak and others 2009a).

# Great Crested Flycatcher

Deanna K. Dawson,
Patuxent Bird Identification InfoCenter
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## Status

The great crested flycatcher (*Myiarchus crinitus*), a neotropical migrant, is found throughout the forests of eastern North America and the riparian habitats of the Mississippi River watershed. Populations have remained relatively stable across most of its range, though in the WGCP they have declined by 1.3 percent per year since 1966 (Sauer and others 2005) (Table 5). This species has a regional combined score of 13 in both the CH and WGCP (Table 1).

## Natural History

The great crested flycatcher is an obligate cavity nester in deciduous forest habitats of the eastern United States; it generally is absent in pure evergreen stands (Lanyon 1997). This species is not area sensitive but does require a minimum amount of forested habitat in the landscape. It may nest in patches as small as 0.2 ha and abundance may decline in forest interiors (Robbins and others 1989). The great crested flycatcher does not occupy riparian corridors surrounded by less than 14.7 percent forest (Perkins and others 2003b), and detection probabilities steadily increase with increasing corridor width (Groom and Grubb 2002).

The great crested flycatcher forages by sallying from exposed perches (Lanyon 1997), so open forest stands are preferred. Holmes and others (2004) found that abundance was highest in heavily cut stands where one-third or more of the basal area was removed. Similarly, Moorman and Guynn (2001) found that the great crested flycatcher was associated with large (0.5 ha) canopy gaps in bottomland hardwood forest in South Carolina. Snags not only provide exposed perches for foraging but also cavities for nesting, and the great crested flycatcher is negatively affected by the removal of snags associated with certain forestry practices (Lohr and others 2002). Where snags are lacking, this species will use nest boxes and other artificial cavities; this enables it to occupy cemeteries, suburban parks, and wooded pastures. Wakeley and Roberts (1996) found that this bird is associated with mesic sites, but this may reflect a preference for bottomland hardwoods over evergreen uplands in the Southeast.

## Model Description

The HSI model for great crested flycatcher includes five variables: landform, landcover, successional age class, snag density, and distance to edge.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 65). We directly assigned SI scores to these combinations on the basis of relative habitat quality associations reported by Hamel (1992) for the great crested flycatcher.

**Table 65.—Relationship of landform, landcover type, and successional age class to suitability index scores for great crested flycatcher habitat; values in parentheses apply to West Gulf Coastal Plain/Ouachitas**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| | Deciduous | 0.000 | 0.333 | 0.333 | 0.500 | 0.667 |
| | Evergreen | 0.000 | 0.333 | 0.333 | 0.500 | 0.667 |
| | Mixed | 0.000 | 0.333 | 0.333 | 0.667 | 1.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| | Woody wetlands | 0.000 | 0.333 | 0.333 | 0.667 | 1.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.333 | 0.333 | 0.583 | 0.834 |
| | Transitional-shrubland | 0.000 | 0.333 | 0.333 | 0.667<br>(0.500) | 1.000<br>(0.667) |
| | Deciduous | 0.000 | 0.333 | 0.333 | 0.583 | 0.834 |
| | Evergreen | 0.000 | 0.333 | 0.333 | 0.500 | 0.667 |
| | Mixed | 0.000 | 0.333 | 0.333 | 0.667 | 1.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.167 | 0.583 | 0.834 |
| | Woody wetlands | 0.000 | 0.333 | 0.333 | 0.667 | 1.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.167 | 0.667 | 1.000 |
| | Transitional-shrubland | 0.000 | 0.333<br>(0.250) | 0.333<br>(0.250) | 0.667<br>(0.500) | 1.000<br>(0.667) |
| | Deciduous | 0.000 | 0.333 | 0.333 | 0.667 | 1.000 |
| | Evergreen | 0.000 | 0.333<br>(0.250) | 0.333<br>(0.250) | 0.500 | 0.667 |
| | Mixed | 0.000 | 0.333 | 0.333 | 0.667 | 1.000 |
| | Orchard-vineyard | 0.000 | 0.333 | 0.333 | 0.667 | 1.000 |
| | Woody wetlands | 0.000 | 0.333 | 0.333 | 0.667 | 1.000 |

The great crested flycatcher relies on snags (SI2) for nesting and foraging. We fit a logistic function (Fig. 35) through average snag values (8.5/ha) observed by Lohr and others (2002), assuming that this value represented average habitat suitability (SI score = 0.500) and that a higher abundance of snags would not be detrimental but increase the likelihood that this bird will use a site (Table 66).

This species is associated with edges (Lanyon 1997), and its abundance declines with increasing distance from an edge (SI3). Small and Hunter (1989) found that more than 60 percent of all flycatchers were less than 60 m from an edge. We assumed maximum habitat suitability at the edge and modeled the relationship between distance to edge and SI score as an inverse logistic function through these data points (Fig. 36, Table 67).

To calculate the overall HSI, we determined the geometric mean of SI scores for forest structure (SI1 and SI2) and then calculated the geometric mean of this value with the edge function (SI3).

$$\text{Overall HSI} = ((\text{SI1} * \text{SI2})^{0.500} * \text{SI3})^{0.500}$$

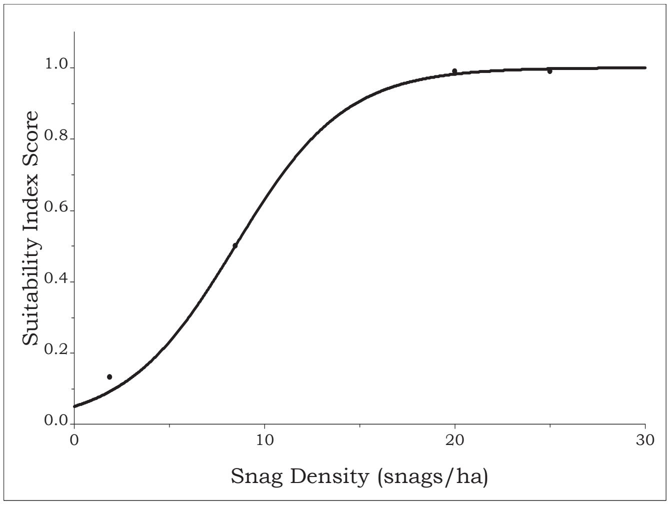


Figure 35.—Relationship between snag density and suitability index (SI) scores for great crested flycatcher habitat. Equation: SI score = 1.001 / (1 + (18.704 * e $^{(-0.346 * \text{snag density})}$)).

**Table 66.—Influence of snag density on suitability index (SI) scores for great crested flycatcher habitat**

| Snag density (snags/ha) | SI score |
|---|---|
| 0.0[a] | 0.000 |
| 1.9[a] | 0.133 |
| 8.5[a] | 0.500 |
| 20.0[b] | 1.000 |
| 25.0[b] | 1.000 |

[a] Lohr and others (2002).
[b] Assumed value.

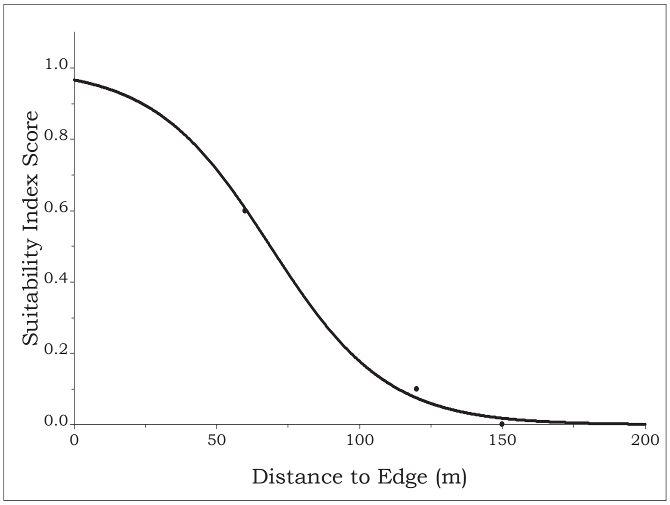


Figure 36.—Relationship between distance to edge and suitability index (SI) scores for great crested flycatcher habitat. Equation: SI score = 1 - (1.000 / (1 + (28.950 * e $^{-0.049 * \text{distance to edge}}$))).

**Table 67.—Influence of distance (m) to edge[a] on suitability index (SI) scores for great crested flycatcher habitat**

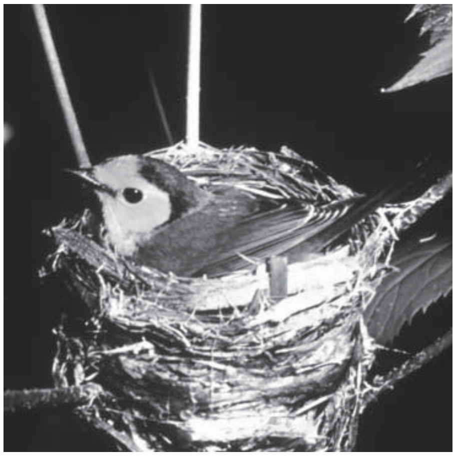
| Distance to edge | SI score |
|---|---|
| 0[b] | 1.0 |
| 60[c] | 0.6 |
| 120[b] | 0.1 |
| 150[b] | 0.0 |

[a] Edge defined by nonhabitat pixels adjacent to habitat pixels (defined by SI1).
[b] Assumed value.
[c] Small and Hunter (1989).

## Verification and Validation

The great crested flycatcher was found in all 88 subsections within the CH and WGCP. Spearman rank correlation on average HSI score and mean BBS route abundance failed to identify a significant ($P \leq 0.001$) association ($r_s = 0.55$) between these two variables. The generalized linear model predicting BBS abundance from BCR and HSI for the great crested flycatcher was not significant ($P = 0.152$; $R^2 = 0.043$), and the coefficient on the HSI predictor variable was negative ($\beta = -2.740$) and not significantly different from zero ($P = 0.151$). Therefore, we considered the HSI model for the great crested flycatcher neither verified nor validated (Tirpak and others 2009a).

## Hooded Warbler



U.S. Fish & Wildlife Service

### Status

The hooded warbler (*Wilsonia citrina*) is a long-distance migrant found throughout the deciduous forests of eastern North America. Because of area sensitivity, it is restricted to forested landscapes and disappears from the forest-prairie ecotone at the western edge of its range faster than other silvicolous species (e.g., eastern wood-pewee). Populations in the WGCP declined prior to 1990 but have since remained stable. Conversely, populations in the CH have increased (Sauer and others 2005) (Table 5). This species is not a Bird of Conservation Concern in either BCR (Table 1) but it is a planning and responsibility species in the WGCP (regional combined score = 16; Table 1). Nearly 30 percent of the continental population of the hooded warbler breeds in the WGCP (Panjabi and others 2001).

### Natural History

The hooded warbler breeds in a variety of habitats, from mixed-hardwood forests in the northern portion of its range to cypress-gum swamps in the South. Regardless of forest type, it prefers mesic sites in large forest tracts (> 15 ha; Evans-Ogden and Stutchbury 1994). Although nest success in small forest patches is not significantly lower than in large patches (Buehler and others 2002), females may avoid small fragments and males use edge less than its availability (Norris and Stutchbury 2002, Norris and others 2000). Occupancy of a site by a nesting pair increases with shrub height and the percentage of vegetation between 1 and 2 m.

This species nests in shrubs within small forest clearings or in the dense understories of closed-canopied forests. As a result, territories often include a mix of open and closed canopies. Gaps created by tree fall or selective logging are particularly attractive (≤ 0.5 ha; Annand and Thompson 1997, Moorman and others 2002, Whittam and others 2002), and the hooded warbler colonizes these sites within 1 to 5 years. Nest sites in Canada had denser ground vegetation, fewer tree stems, lower basal area of small trees, and greater basal area of large trees than control sites (Whittam and others 2002). Bisson and Stutchbury (2000) concluded that canopy gaps and density of understory vegetation were the most important factors affecting site selection. Repeated burning, which removed understory vegetation, reduced hooded warbler abundance in Ohio (Artman and others 2001). This species is a common cowbird host, which may explain its sensitivity to fragmentation (Donovan and Flather 2002).

### Model Description

The HSI model for the hooded warbler includes seven variables: landform, land cover, successional age class, small stem (< 2.5 cm d.b.h.) density, canopy cover, forest patch size, and percent forest in a 1-km landscape.

**Table 68.—Relationship of landform, landcover type, and successional age class to suitability index scores for hooded warbler habitat; values in parentheses apply to West Gulf Coastal Plain/Ouachitas**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.167 | 0.667 | 1.000 |
| | Deciduous | 0.000 | 0.000 | 0.167 | 0.667 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.334 | 0.667 |
| | Mixed | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.167 | 0.667 | 1.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.167<br>(0.000) | 0.500<br>(0.334) | 0.667 |
| | Deciduous | 0.000 | 0.000 | 0.167 | 0.667 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.334 | 0.667 |
| | Mixed | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.167 | 0.667 | 1.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.167<br>(0.000) | 0.500<br>(0.167) | 0.667<br>(0.334) |
| | Deciduous | 0.000 | 0.000 | 0.167 | 0.667 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.334<br>(0.167) | 0.667<br>(0.334) |
| | Mixed | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.167 | 0.667 | 1.000 |

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 68). We directly assigned SI scores to these combinations on the basis of relative habitat quality rankings from Hamel (1992) for the hooded warbler in the Southeast.

This species occupies dense understories in mature forested habitats, so we included both small stem density (SI2) and canopy cover (SI3) in our model. We fit a logistic function (Fig. 37) that links small stem density to SI scores on the basis of data from Annand and Thompson (1992) and Moorman and others (2002) (Table 69). We assumed that the average stem density measured at nest sites by Moorman and others (2002) (4,700 stems/ha) was representative of ideal habitat conditions for the hooded warbler and that there was no upper threshold above which habitat suitability declined. We also fit a logistic function (Fig. 38) to data from Annand and Thompson (1997) (Table 70) to link canopy cover values to SI scores.

We included forest patch size (SI4) as a model predictor because of the negative effect of fragmentation on this species. We used an exponential curve (Fig. 39) to predict habitat

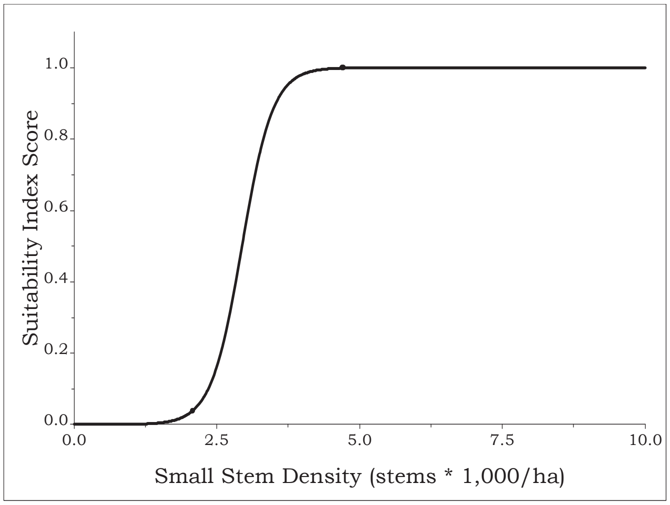


Figure 37.—Relationship between small stem (< 2.5 cm d.b.h.) density (stems * 1000/ha) on suitability index (SI) scores for hooded warbler habitat. Equation: SI score = 1.000 / (1.000 + (102634.340 * e $^{-4.017 * \text{(small stem density / 1000)}}$)).

**Table 69.—Influence of small stem (< 2.5 cm d.b.h.) density (stems * 1,000/ha) on suitability index (SI) scores for hooded warbler habitat**

| Small stem density | SI score |
|---|---|
| 0.000[a] | 0.000 |
| 2.077[b] | 0.039 |
| 4.700[c] | 1.000 |
| 4.717[b] | 1.000 |
| 10.000[a] | 1.000 |

[a] Assumed value.
[b] Annand and Thompson (1992).
[c] Moorman and others (2002).

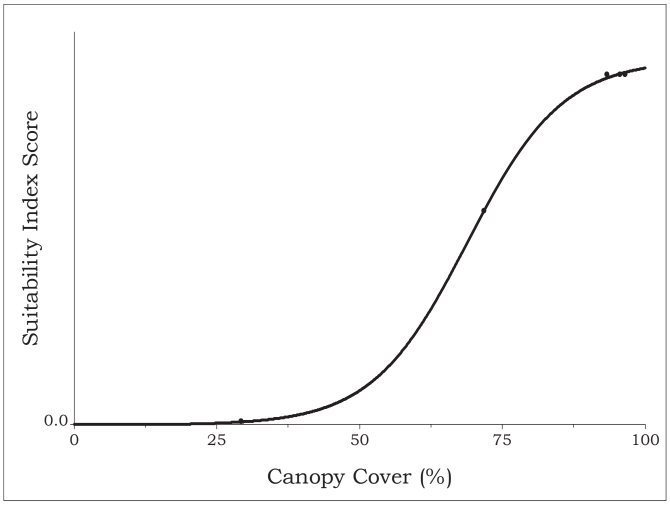


Figure 38.—Relationship between canopy cover on suitability index (SI) scores for hooded warbler habitat. Equation: SI score = 1.024 / (1.000 + (3823.776 * e $^{-0.120 * \text{canopy cover}}$)).

**Table 70.—Influence of canopy cover on suitability index (SI) scores for hooded warbler habitat**

| Canopy cover (percent) | SI score |
|---|---|
| 0.00[a] | 0.0 |
| 29.26[b] | 0.0 |
| 71.86[b] | 0.6 |
| 93.38[b] | 1.0 |
| 95.58[b] | 1.0 |
| 96.59[b] | 1.0 |

[a] Assumed value.
[b] Annand and Thompson (1997).

suitability from forest patch size on the basis of data from Evans-Ogden and Stutchbury (1994) and Kilgo and others (1998). To convert riparian widths reported by Kilgo and others (1998) to forest patch sizes, we assumed that all riparian strips were 10 km long (Table 71). The suitability of a specific forest patch is influenced by the percentage of forest in the landscape (SI5). Small patches that otherwise would be unsuitable may be occupied when in close proximity to a large forest block or in a predominantly forested landscape (Rosenberg and others 1999). To capture this relationship, we fit a logistic function (Fig. 40) to data (Table 72) derived from Donovan and others (1997), who observed differences in predator and brood parasite communities among highly fragmented (< 15 percent), moderately fragmented (45 to 50 percent), and lightly fragmented (> 90 percent forest) landscapes. We assumed that the midpoints between these classes (30 and 70 percent

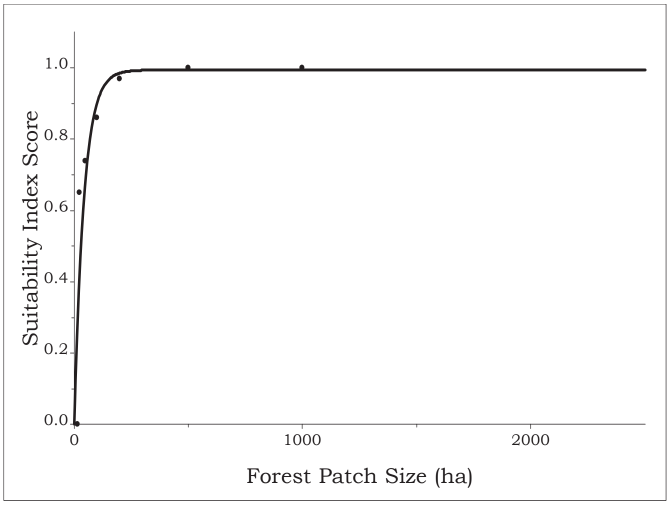


Figure 39.—Relationship between forest patch size and suitability index (SI) scores for hooded warbler habitat. Equation: SI score = $0.994 * (1 - e^{-0.024 * \text{forest patch size}})$.

**Table 71.—Influence of forest patch size on suitability index (SI) scores for hooded warbler habitat**

| Forest patch size (ha) | SI score |
|---|---|
| 15[a] | 0.00 |
| 25[b] | 0.65 |
| 50[b] | 0.74 |
| 100[b] | 0.86 |
| 200[b] | 0.97 |
| 500[b] | 1.00 |
| 1,000[b] | 1.00 |
| 2,500[b] | 1.00 |

[a] Evans-Ogden and Stutchbury (1994).

[b] Kilgo and others (1998).

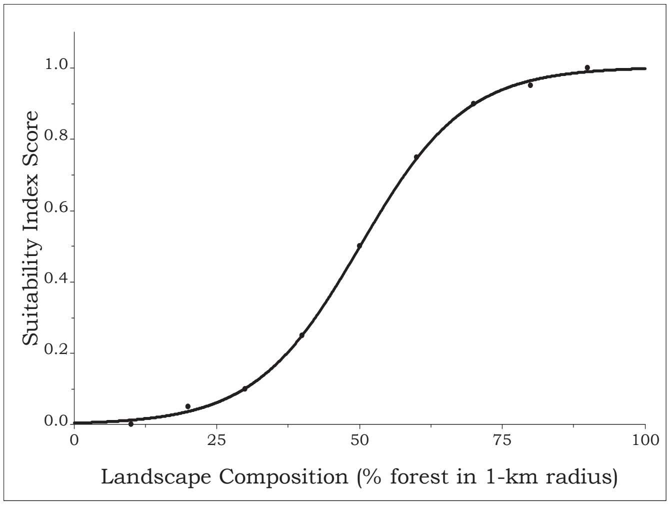


Figure 40.—Relationship between landscape composition and suitability index (SI) scores for hooded warbler habitat. Equation: SI score = $1.005 / (1.000 + (221.816 * e^{-0.108 * (\text{landscape composition})}))$.

**Table 72.—Relationship between landscape composition (percent forest in 1-km radius) and suitability index (SI) scores for hooded warbler habitat**

| Landscape composition | SI score |
|---|---|
| 0[a] | 0.00 |
| 10[a] | 0.00 |
| 20[a] | 0.05 |
| 30[b] | 0.10 |
| 40[a] | 0.25 |
| 50[b] | 0.50 |
| 60[a] | 0.75 |
| 70[b] | 0.90 |
| 80[a] | 0.95 |
| 90[b] | 1.00 |
| 100[a] | 1.00 |

[a] Assumed value.

[b] Dononvan and others (1997).

forest) defined the specific cutoffs for poor (SI score ≤ 0.10) and excellent (SI score ≥ 0.90) habitat, respectively. We used the maximum SI score from SI4 or SI5 to account for the higher suitability of small forest patches in a heavily forested landscape.

The overall HSI score was calculated as the geometric mean of the geometric mean of the SI values from the landform, landcover, and successional age class matrix, small stem density, and canopy cover functions (SI1, SI2, and SI3) multiplied by the maximum value of either the forest patch size or percent forest in the 1-km radius landscape functions (SI4 and SI5).

$$\text{Overall HSI} = ((\text{SI1} * \text{SI2} * \text{SI3})^{0.333} * \text{Max}(\text{SI4 or SI5}))^{0.500}$$

## Verification and Validation

The hooded warbler was found in 84 of the 88 subsections within the CH and WGCP. Spearman rank correlations identified significant positive associations between average HSI score and mean BBS route abundance across all subsections ($P \leq 0.001$; $r_s = 0.49$) and subsections within which this species was detected ($P \leq 0.001$; $r_s = 0.42$). The generalized linear model predicting BBS abundance from BCR and HSI for the hooded warbler was significant ($P \leq 0.001$; $R^2 = 0.551$), and the coefficient on the HSI predictor variable was both positive ($\beta = 8.190$) and significantly different from zero ($P \leq 0.001$). Therefore, we considered the HSI model for the hooded warbler both verified and validated (Tirpak and others 2009a).

## Kentucky Warbler


U.S. Fish & Wildlife Service

### Status

The Kentucky warbler (*Oporornis formosus*) breeds throughout the southeastern United States; densities are highest west of the Appalachian front. Populations have been stable in the CH over the last 40 years, but have declined in the WGCP by 2.2 percent per year during this period (Table 5). This species requires management attention in both regions (regional combined score = 18 and 19 in the CH and WGCP, respectively). A high percentage of the continental population breeds in both BCRs (28 and 22 percent, respectively; Panjabi and others 2001). The species is an FWS Bird of Conservation Concern in the WGCP (Table 1).

### Natural History

The Kentucky warbler, a long-distance migrant, breeds in mature moist deciduous forests of the Southeast. It is a forest-interior specialist, primarily because of low productivity and survival in edge and early successional habitats (Morse and Robinson 1999; Robinson and Robinson 2001). The Kentucky warbler occupies fragments as small as 2.4 ha (Blake and Karr 1987) but tracts larger than 500 ha are considered the minimum size necessary to support sustainable populations (McDonald 1998). A dense understory is a common feature of nesting sites. Ground cover averaged 46 percent in Kentucky warbler territories in Missouri (Wenny and others 1993), and vegetation of less than 1.5 m was denser around nests than random sites in South Carolina (Kilgo and others 1996). Dense vegetation (0.3 to 1 m) was also associated with higher numbers of the Kentucky warbler in Maryland (Robbins and others 1989). Mesic sites are universally selected (McShea and others 1995, McDonald 1998, Gram and others 2003).

### Model Description

The habitat suitability model for the Kentucky warbler includes six variables: landform, landcover, successional age class, small stem (< 2.5 cm d.b.h.) density, forest patch size, and percent forest in the landscape.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 73). We relied on relative habitat quality associations reported by Hamel (1992) to assign SI scores to these combinations. However, we increased SI scores for shrub-seedling stands on the basis of data from Thompson and others (1992).

The Kentucky warbler nests at the base of shrubs and occupies habitats containing high densities of small stems (SI2). We used data on the relative abundance of this species from Wenny and others (1993), Kilgo and others (1996), and Annand and Thompson (1997) to derive a logistic function (Fig. 41) that predicts habitat suitability from small stem density (Table 74).

**Table 73.—Relationship of landform, landcover type, and successional age class to suitability index scores for Kentucky warbler habitat; values in parentheses apply to West Gulf Coastal Plain/Ouachitas**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.667 | 0.417 | 0.667 | 0.667 |
| | Deciduous | 0.000 | 0.667 | 0.417 | 0.667 | 0.667 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.333 | 0.167 | 0.333 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 1.000 | 0.667 | 1.000 | 1.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.333<br>(0.000) | 0.167<br>(0.000) | 0.333<br>(0.000) | 0.333<br>(0.000) |
| | Deciduous | 0.000 | 0.667 | 0.334 | 0.667 | 0.667 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.333 | 0.167 | 0.333 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 1.000 | 0.667 | 1.000 | 1.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.333<br>(0.000) | 0.167<br>(0.000) | 0.333<br>(0.000) | 0.333<br>(0.000) |
| | Deciduous | 0.000 | 0.500 | 0.250 | 0.500 | 0.500 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.333 | 0.167 | 0.333 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 1.000 | 0.667 | 1.000 | 1.000 |

We used a logarithmic function (Fig. 42) to quantify the relationship between forest patch size (SI3) and habitat suitability on the basis of minimum patch size observations by Hayden and others (1985) and occupancy rates in different patch sizes reported by Robbins and others (1989) (Table 75). However, the suitability of a specific forest patch is influenced by its landscape context (SI4). Because the Kentucky warbler is particularly sensitive to fragmentation (Lynch and Whigham 1984), we used a 10-km window to characterize the landscape. We fit a logistic function (Fig. 43) to data (Table 76) derived from Donovan and others (1997), who observed differences in predator and brood parasite communities among highly fragmented (< 15 percent), moderately fragmented (45 to 50 percent), and lightly fragmented (> 90 percent forest) landscapes. We assumed that the midpoints between these classes (30 and 70 percent forest) defined the specific cutoffs for poor (SI score ≤ 0.10) and excellent (SI score ≥ 0.90) habitat, respectively.

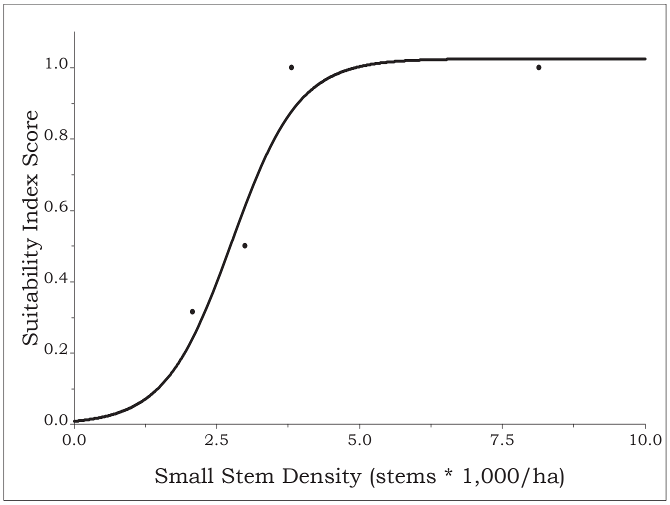


Figure 41.—Relationship between small stem (< 2.5 cm d.b.h.) density (stems * 1000/ha) and suitability index (SI) scores for Kentucky warbler habitat. Equation: SI score = 1.026 / (1.000 + (111.558 * e -1.707 * $^{\text{(small stem density / 1000)}}$)).

**Table 74.—Influence of small stem (< 2.5 cm d.b.h.) density (stems/ha) on suitability index (SI) scores for Kentucky warbler habitat**

| Small stem density | SI score |
|---|---|
| 0.000[a] | 0.000 |
| 2.077[b] | 0.316 |
| 3.000[c] | 0.500 |
| 3.812[b] | 1.000 |
| 8.148[b] | 1.000 |
| 47.600[d] | 1.000 |

[a] Assumed value.

[b] Annand and Thompson (1997).

[c] Wenny and others (1993).

[d] Kilgo and others (1996).

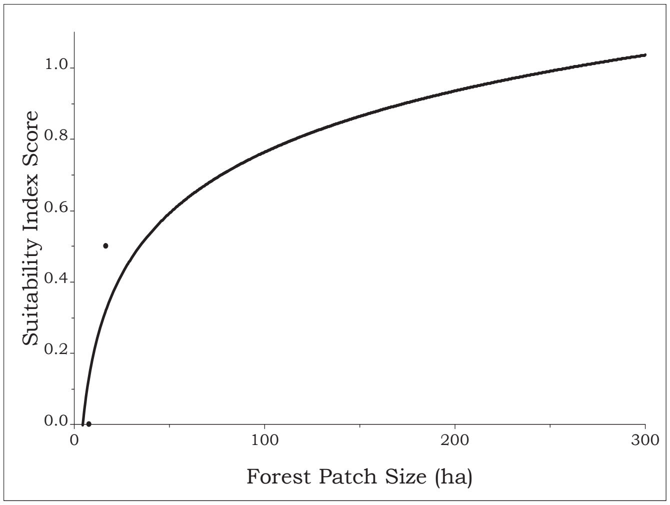


Figure 42.—Relationship between forest patch size and suitability (SI) scores for Kentucky warbler habitat. Equation: SI score = 0.248 * ln(forest patch size) – 0.377.

**Table 75.—Influence of forest patch size on suitability index (SI) scores for Kentucky warbler habitat**

| Forest patch size (ha) | SI score |
|---|---|
| 8[a] | 0.0 |
| 17[b] | 0.5 |
| 300[b] | 1.0 |

[a] Hayden and others (1985).

[b] Robbins and others (1989).

To calculate the overall HSI score, we determined the geometric mean of SI scores for functions relating to forest structure (SI1 and SI2) and landscape composition (SI3 and SI4) separately and then the geometric mean of these means together.

Overall HSI = $((SI1 * SI2)^{0.500} * (SI3 * SI4)^{0.500})^{0.500}$

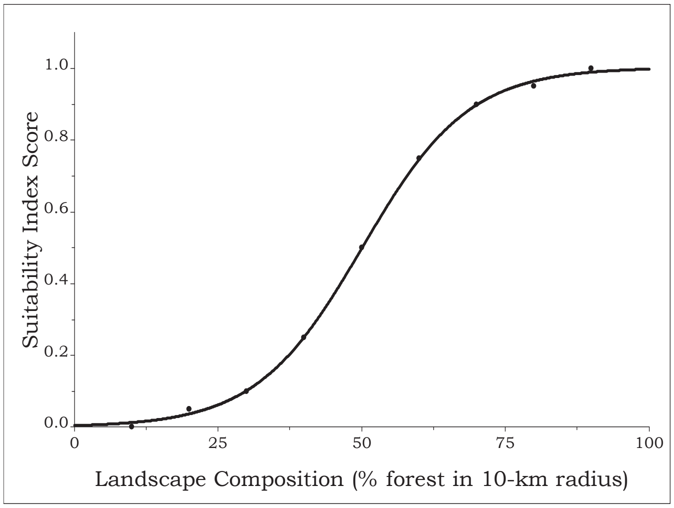


Figure 43.—Relationship between landscape composition and suitability index (SI) scores for Kentucky warbler habitat. Equation: SI score = 1.005 / (1.000 + (221.816 * e $^{-0.108\ *\ (\text{landscape composition})}$)).

**Table 76.—Relationship between landscape composition (percent forest in 10-km radius) and suitability index (SI) scores for Kentucky warbler habitat**

| Landscape composition | SI score |
|---|---|
| 0[a] | 0.00 |
| 10[a] | 0.00 |
| 20[a] | 0.05 |
| 30[b] | 0.10 |
| 40[a] | 0.25 |
| 50[b] | 0.50 |
| 60[a] | 0.75 |
| 70[b] | 0.90 |
| 80[a] | 0.95 |
| 90[b] | 1.00 |
| 100[a] | 1.00 |

[a] Assumed value.

[b] Dononvan and others (1997).

## Verification and Validation

The Kentucky warbler was found in all 88 subsections of the CH and WGCP. Spearman rank correlations identified a significant positive association between average HSI score and mean BBS route abundance across all subsections ($P \leq 0.001$; $r_s = 0.71$). The generalized linear model predicting BBS abundance from BCR and HSI for the Kentucky warbler was significant ($P \leq 0.001$; $R^2 = 0.346$), and the coefficient on the HSI predictor variable was both positive ($\beta = 6.351$) and significantly different from zero ($P \leq 0.001$). Therefore, we considered the HSI model for the Kentucky warbler both verified and validated (Tirpak and others 2009a).

## Louisiana Waterthrush


Charles H. Warren, images.nbii.gov

### Status

The Louisiana waterthrush (*Seirus motacilla*) is a long-distance neotropical migrant found throughout the deciduous forests of the eastern and central United States. The small population in the WGCP has remained relatively stable since 1966 while the larger population in the CH has increased by 2.6 percent annually (Sauer and others 2005) (Table 5). This species is a Bird of Conservation Concern in both regions (Table 1). However, PIF differentiates the priority for this species in the CH (planning and responsibility, regional combined score = 15) and WGCP (management attention, regional combined score = 18; Table 1).

### Natural History

As its name implies, the Louisiana waterthrush is associated with water throughout its range (Robinson 1995). Densities are highest along gravel-bottomed, first- and second-order streams flowing through large (> 350 ha) tracts of mature deciduous forest (Robbins and others 1989, Robinson 1995). Birds also breed at lower densities along mud-bottomed streams in cypress swamps and bottomland hardwood forests (Hamel 1992, Robinson 1995).

Prosser and Brooks (1998) developed and validated an HSI model for the Louisiana waterthrush in central Pennsylvania that included eight variables: canopy cover (> 80 percent considered ideal), shrub cover (< 25 percent), ratio of deciduous to conifer cover (30 to 69 percent, mostly reflecting hemlock dominance along streams in the Northeast), herbaceous cover (< 25 percent), stream order (first- or second-order with well developed pools and riffles), water clarity and substrate (clear and rocky or sandy), nesting cover (presence of uprooted trees or creviced, steep banks), and forest area (> 350 ha).

### Model Description

Our HSI model for the Louisiana waterthrush included eight variables: landform, landcover, successional age class, distance to stream, canopy cover, small stem (< 2.5 cm d.b.h.) density, forest patch size, and percent forest in a 1-km radius.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 77). We directly assigned SI scores to these combinations on the basis of vegetation and successional age class associations outlined in Hamel (1992).

We included distance to stream (SI2) as a variable because the waterthrush uses streams and creeks for foraging and nesting. The Louisiana waterthrush restricts its foraging to the streambed and bank, so we assumed a sharp decline in suitability with increasing distance to a stream (Table 78). We used an inverse logistic function to characterize this relationship (Fig. 44).

**Table 77.—Relationship of landform, landcover type, and successional age class to suitability index (SI) scores for Louisiana waterthrush habitat**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.500 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.167 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.500 | 1.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.500 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.167 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.500 | 1.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.250 | 0.500 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.167 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.334 | 0.667 |

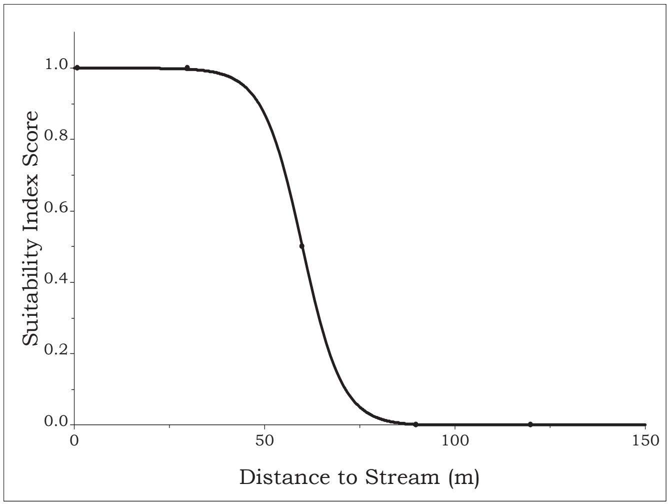


Figure 44.—Relationship between distance to stream and suitability index (SI) scores for Louisiana waterthrush habitat. Equation: SI score = 1 - (1.0015 / (1 + (104411.5 * e $^{-0.1926 * \text{distance to stream}}$))).

**Table 78.—Relationship between distance to stream and suitability index (SI) scores for Louisiana waterthrush habitat.**

| Distance to stream (m)[a] | SI score |
|---|---|
| 0 | 1.0 |
| 30 | 1.0 |
| 60 | 0.5 |
| 90 | 0.0 |
| 120 | 0.0 |

[a] Assumed value.

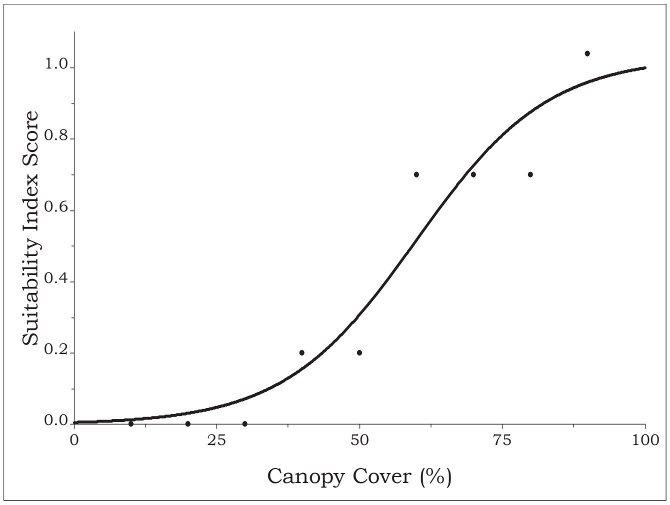


Figure 45.—Relationship between canopy cover and suitability index (SI) scores for Louisiana waterthrush habitat. Equation: SI score = (1.0313 / (1 + (175.8083 * $e^{-0.0864 * \text{canopy cover}}$))).

**Table 79.—Relationship between canopy cover and suitability index (SI) scores for Louisiana waterthrush habitat**

| Canopy cover (percent)[a] | SI score |
|---|---|
| 0 | 0.0 |
| 10 | 0.0 |
| 20 | 0.0 |
| 30 | 0.0 |
| 40 | 0.2 |
| 50 | 0.2 |
| 60 | 0.7 |
| 70 | 0.7 |
| 80 | 0.7 |
| 90 | 1.0 |

[a] Prosser and Brooks (1998).

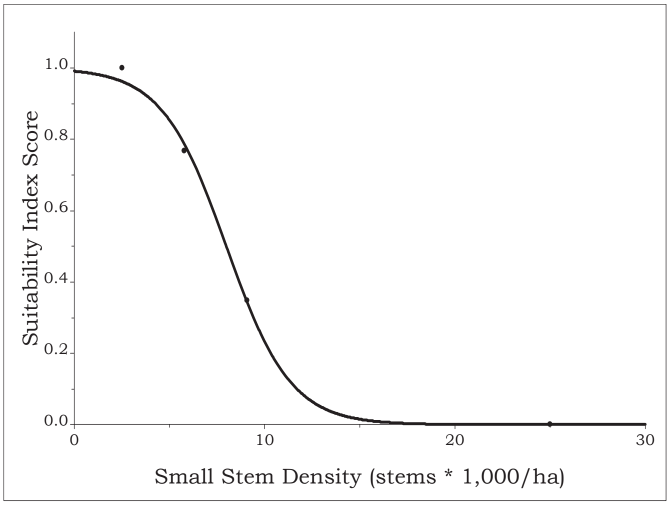


Figure 46.—Relationship between small stem (< 2.5 cm d.b.h.) density (stems * 1000/ha) and suitability index (SI) scores for Louisiana waterthrush habitat. Equation: SI score = 1 - (1.000 / (1 + (113.261 * $e^{-0.592 * (\text{small stem density} / 1000)}$))).

**Table 80.—Relationship between small stem (< 2.5 cm d.b.h.) density (stems * 1,000/ha) and suitability index (SI) scores for Louisiana waterthrush habitat**

| Small stem density | SI score |
|---|---|
| 0[a] | 1.000 |
| 2.519[a] | 1.000 |
| 5.803[a] | 0.767 |
| 9.086[a] | 0.349 |
| 25.000[b] | 0.000 |

[a] Prosser and Brooks (1998).
[b] Assumed value.

We also included canopy cover (SI3) and small stem density (SI4) as variables based on the preference of this species for mature forested sites with closed canopies and open understories. We fit logistic (Fig. 45) and inverse logistic (Fig. 46) functions to data adapted from the HSI model of Prosser and Brooks (1998) for canopy cover (Table 79) and small stem density (Table 80), respectively.

Forest patch size (SI5) affects the occupancy of habitats by the Louisiana waterthrush. To predict habitat suitability from forest patch size, we fit a logarithmic function (Fig. 47) to data from Hayden and others (1985) and Robbins and others (1989) (Table 81)

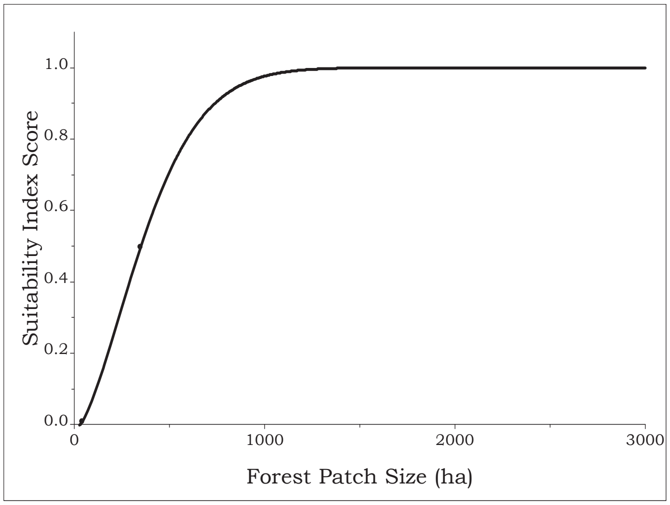


Figure 47.—Relationship between forest patch size and suitability index (SI) scores for Louisiana waterthrush habitat. Equation: SI score = 1.000 – (1.010 * e $^{-0.0003 * (\text{forest patch size} \wedge 1.321)}$).

**Table 81.—Relationship between forest patch size and suitability index (SI) scores for Louisiana waterthrush habitat**

| Forest patch size (ha) | SI score |
|---|---|
| 42.2[a] | 0.0 |
| 350[b] | 0.5 |
| 3,200[b] | 1.0 |

[a]Hayden and others (1985).

[b]Robbins and others (1989).

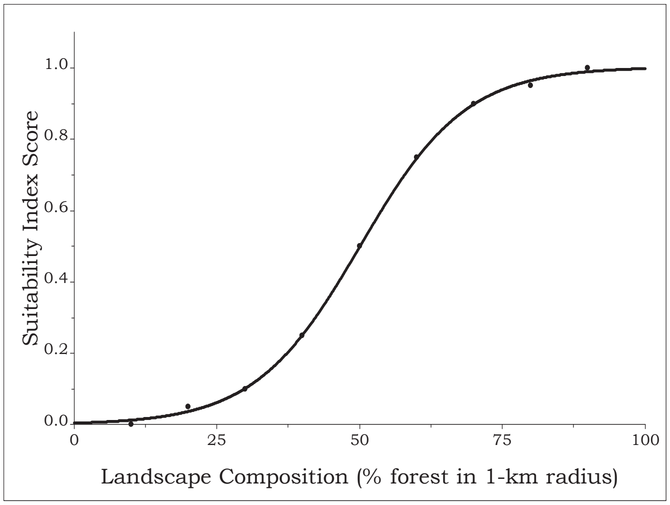


Figure 48.—Relationship between landscape composition and suitability index (SI) scores for Louisiana waterthrush habitat. Equation: SI score = 1.005 / (1.000 + (221.816 * e $^{-0.108 * (\text{landscape composition})}$)).

**Table 82.—Relationship between landscape composition (percent forest in 1-km radius) and suitability index (SI) scores for Louisiana waterthrush habitat**

| Landscape composition | SI score |
|---|---|
| 0[a] | 0.00 |
| 10[a] | 0.00 |
| 20[a] | 0.05 |
| 30[b] | 0.10 |
| 40[a] | 0.25 |
| 50[b] | 0.50 |
| 60[a] | 0.75 |
| 70[b] | 0.90 |
| 80[a] | 0.95 |
| 90[b] | 1.00 |
| 100[a] | 1.00 |

[a]Assumed value.

[b]Donovan and others (1997).

on the detection probabilities of the Louisiana waterthrush in patches of varying size. However, forest patch size alone may not be an appropriate measure of a site's suitability. In predominantly forested landscapes, small patches otherwise not suitable may be occupied due to their proximity to large forest blocks (Rosenberg and others 1999). To capture this relationship, we fit a logistic function (Fig. 48) to data (Table 82) derived from Donovan and others (1997), who observed differences in predator and brood parasite communities among highly fragmented (< 15 percent), moderately fragmented (45 to 50 percent), and lightly fragmented (> 90 percent forest) landscapes. We assumed the midpoints between these classes (30 and 70 percent forest) defined the specific cutoffs for poor (SI score ≤ 0.10) and excellent (SI score ≥ 0.90) habitat, respectively. We used the maximum SI score from

SI5 or SI6 to ensure that small forest blocks in predominantly forested landscapes were assigned an appropriate suitability score.

To calculate the overall HSI, we determined the geometric mean of SI scores for forest structure (SI1, SI3, and SI4) and landscape composition (Max (SI5 or SI6) and SI2) separately and then the geometric mean of these means together.

Overall HSI = $((\text{SI1} * \text{SI3} * \text{SI4})^{0.333} * (\text{Max (SI5 or SI6)} * \text{SI2})^{0.500})^{0.500}$

### Verification and Validation

The Louisiana waterthrush was found in all 88 subsections of the CH and WGCP. Spearman rank correlation on average HSI score and mean BBS route abundance per subsection identified a significant ($P \leq 0.001$) positive association ($r_s = 0.56$) between these two variables. The generalized linear model predicting BBS abundance from BCR and HSI for the Louisiana waterthrush was significant ($P \leq 0.001$; $R^2 = 0.263$), and the coefficient on the HSI predictor variable was both positive ($\beta = 3.664$) and significantly different from zero ($P \leq 0.001$). Therefore, we considered the HSI model for the Louisiana waterthrush both verified and validated (Tirpak and others 2009a).

## Mississippi Kite

Peter S. Weber, www.wildbirdphotos.com
Photo used with permission

### Status

The Mississippi kite (*Ictinia mississippiensis*), a neotropical migrant raptor, is restricted to the Coastal Plains as well as the lower Mississippi and Red River Valleys. Like many birds of prey, this species has exhibited dramatic recoveries over the last 25 years from historical lows in the 1970s. However, its general scarcity prevents BBS from detecting statistically significant trends (Sauer and others 2005; Table 5). The Mississippi kite is not a Bird of Conservation Concern in the CH or WGCP (Table 1). It has a regional combined score of 14 in the CH and 16 in the WGCP.

### Natural history

The Mississippi kite exhibits two breeding strategies within its range. In the southern Great Plains, it is a colonial nester that often inhabits urban areas. In the Mississippi Valley and farther east, this bird is less colonial and nests singly in large trees in bottomland forest and riparian woodlands. Nests from birds within the eastern population generally are located in large (> 22 ha) unfragmented forest near open habitats where birds forage aerially (Parker 1999).

### Model Description

The HSI model for the Mississippi kite includes six variables: landform, land cover, successional age class, forest patch size, interspersion of forest and open habitats, and density of dominant trees.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 83). We directly assigned SI scores to these combinations on the basis of relative habitat quality ranks reported by Hamel (1992) for this species. However, we restricted the Mississippi kite to sawtimber stands based on its preference for mature forest stands (Parker 1999).

We also included forest patch size (SI2) in the model and used the range and mean of patch sizes reported by Barber and others (1998) to define the minimum, maximum, and average patch sizes associated with nonhabitat, optimal, and average habitat suitability for this function, respectively (Table 84; Fig. 49).

The Mississippi kite requires large patches of forest and grassland in a specific landscape context (Parker 1999, Coppedge and others 2001). We used the relative amount of these habitats within a 1-km radius as an index to their interspersion at the landscape scale (SI3). We assumed that habitat suitability was optimal in open habitats with few trees (70 to 90 percent agriculture or grassland) or landscapes containing moderate forest cover interspersed with open habitats (60 to 70 percent forest; Table 85).

**Table 83.—Relationship of landform, landcover type, and successional age class to suitability index scores for Mississippi kite habitat. Values in parentheses apply to West Gulf Coastal Plain/Ouachitas.**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.500 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.500 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.000 | 0.500 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.333 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.500 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 1.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.500 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.333 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.000 | 0.500 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.333 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.500 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 1.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.500 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.333<br>(0.167) |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.000 | 0.500 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.333<br>(0.167) |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.500 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 1.000 |

The Mississippi kite nests in dominant trees (SI4) that extend above the canopy. Parker (1999) identified old-growth stands and isolated trees as preferred nesting substrates for this species, and Barber and others (1998) observed the Mississippi kite using nest trees that were higher and larger in d.b.h. than those in the surrounding overstory. We assumed that a tree with a d.b.h. greater than 76.2 cm in a sawtimber stand would extend above the canopy and provide an adequate nest substrate for this species. We further assumed that one dominant tree per ha would satisfy this requirement and that the Mississippi kite would be absent from stands with a uniform canopy (zero dominant trees/ha). We fit an exponential function (Fig. 50) to the values between these data points. Stands with 14 dominant trees per ha (the maximum observed in the WGCP during the FIA surveys of the 1990s) were associated with maximum habitat suitability (Table 86).

To calculate the overall HSI score, we determined the geometric mean of SI scores for forest structure (SI1 and SI4) and landscape composition (SI2 and SI3) separately and then the geometric mean of these means together.

Overall HSI = $((\text{SI1} * \text{SI4})^{0.500} * (\text{SI2} * \text{SI3})^{0.500})^{0.500}$

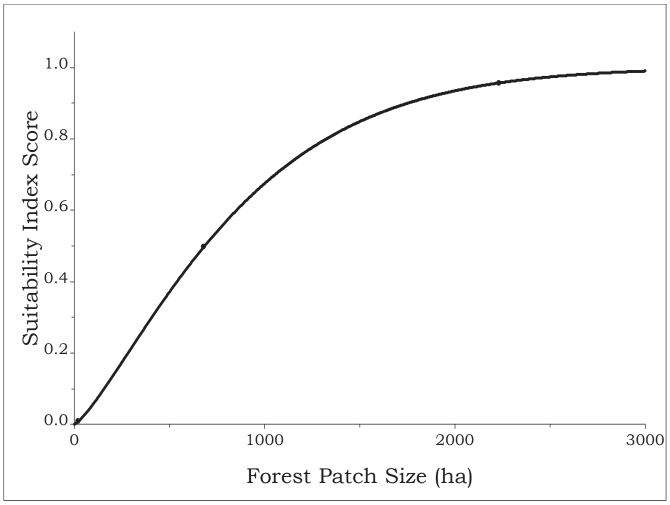


Figure 49.—Relationship between forest patch size and suitability index (SI) scores for Mississippi kite habitat. Equation: SI score = 1.002 – (1.000 * e $^{-0.0002 * (\text{forest patch size} \wedge 1.278)}$).

**Table 84.—Influence of forest patch size on suitability index (SI) scores for Mississippi kite habitat**

| Forest patch size (ha)[a] | SI score |
|---|---|
| 22 | 0.0 |
| 683 | 0.5 |
| 3,000 | 1.0 |

[a] Barber and others (1998).

**Table 85.—Suitability index scores for Mississippi kite habitat based on proportion of cells providing roosting and nesting habitat within 1-km radius**

| | Proportion forest[a] | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Proportion agriculture-grassland[b] | 0.0 | 0.1 | 0.2 | 0.3 | 0.4 | 0.5 | 0.6 | 0.7 | 0.8 | 0.9 | 1.0 |
| 0.0 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.50 | 0.50 | 0.50 | 0.50 |
| 0.1 | 0.00 | 0.00 | 0.00 | 0.00 | 0.20 | 0.20 | 0.40 | 0.60 | 0.60 | 0.60 | |
| 0.2 | 0.00 | 0.00 | 0.00 | 0.00 | 0.40 | 0.40 | 0.60 | 0.80 | 0.80 | | |
| 0.3 | 0.00 | 0.00 | 0.00 | 0.00 | 0.60 | 0.60 | 0.80 | 1.00 | | | |
| 0.4 | 0.35 | 0.40 | 0.40 | 0.60 | 0.80 | 0.80 | 1.00 | | | | |
| 0.5 | 0.50 | 0.50 | 0.55 | 0.70 | 0.70 | 0.60 | | | | | |
| 0.6 | 0.60 | 0.70 | 0.75 | 0.90 | 0.80 | | | | | | |
| 0.7 | 0.70 | 0.75 | 1.00 | 1.00 | | | | | | | |
| 0.8 | 0.80 | 0.90 | 1.00 | | | | | | | | |
| 0.9 | 0.80 | 1.00 | | | | | | | | | |
| 1.0 | 0.80 | | | | | | | | | | |

[a] Woody wetlands, deciduous forest, low-density residential.

[b] Open water, open fields (natural or cultivated), emergent herbaceous wetland.

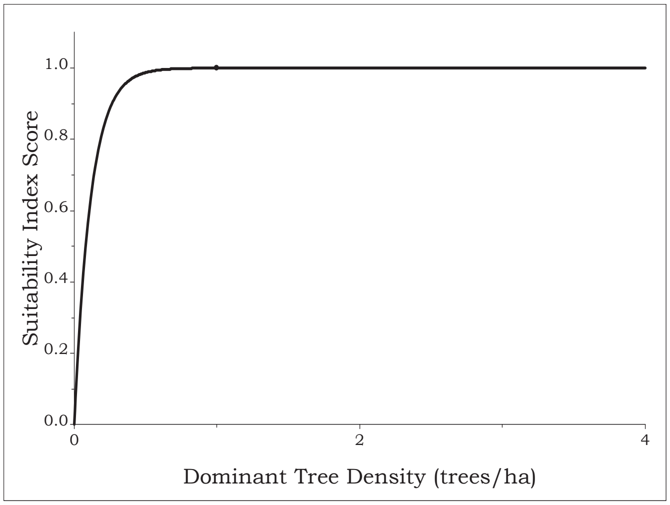


Figure 50.—Relationship between dominant tree (> 76.2 cm d.b.h.) density and suitability index (SI) scores for Mississippi kite habitat. Equation: SI score = $1 - e^{-8.734 * \text{dominant tree density}}$.

**Table 86.—Influence of dominant tree (d.b.h. > 76.2 cm) density (trees/ha) on suitability index (SI) scores for Mississippi kite habitat**

| Dominant tree density[a] | SI score |
|---|---|
| 0 | 0.0 |
| 1 | 1.0 |
| 14 | 1.0 |

[a] Assumed value.

## Verification and Validation

The Mississippi kite was found in 49 of the 88 subsections within the CH and WGCP. Spearman rank correlations based on all subsections yielded a significant ($P$ = 0.003) positive association ($r_s$ = 0.31) between average HSI score and mean BBS route abundance. However, this association was not evident when the correlation considered only subsections in which this species was found. The generalized linear model predicting BBS abundance from BCR and HSI for the Mississippi kite was significant ($P \leq 0.001$; $R^2$ = 0.287); however, the coefficient on the HSI predictor variable was negative (ß = -0.176). Therefore, we considered the HSI model for the Mississippi kite verified but not validated (Tirpak and others 2009a).

## Northern Bobwhite

U.S. Forest Service

### Status

The northern bobwhite (*Colinus virginianus*) is a resident gamebird found throughout the eastern United States and Great Plains. Populations have declined by 3 percent per year since 1966 (Sauer and others 2005). Declines in the CH and WGCP have been equally dramatic (3.1 and 4.4 percent per year, respectively) during this period (Table 5). As a resident gamebird, this species is not afforded special status by the FWS (protection is relegated to state wildlife agencies). Nevertheless, PIF has designated this bird as one requiring management attention in both the CH and WGCP (regional combined scores = 16 and 15, respectively) (Table 1). To address rangewide declines in populations, the Northern Bobwhite Conservation Initiative was established in 2002.

### Natural History

The northern bobwhite is an economically important gamebird in the southern and central United States (Brennan 1999). It is associated with early successional vegetation, making use of agricultural fields, grasslands, grass-shrub rangelands, park-like pine forests and mixed pine-hardwood forests. At the county scale in Texas, the area in cultivated land and livestock density show curvilinear relationships to bobwhite population indices (Lusk and others 2002a). In Oklahoma, bobwhite indices decrease with the proportion of the landscape in mature woodland, but increase with the proportion of brushy prairie or early successional habitat (Guthery and others 2001). Guthery and others (2001) found that populations were highest in areas lacking cropland agriculture. However, Williams and others (2000) found that the bobwhite selected cropland when it accounted for a small proportion of the landscape. Patterns of use and survival differ between crop-dominated and rangeland-dominated areas during the hunting season in Kansas (Williams and others 2000). Bobwhite densities vary across the range depending on habitat quality but are highest in areas with small (0.5 to 5.0 ha) interspersed patches of habitat.

Frequency and intensity of disturbance are important for this species, especially in southern pine forests where prescribed burning is a useful management tool. Cram and others (2002) reported higher bobwhite abundance in pine-grassland restoration areas in Arkansas as conifer and hardwood basal area decreased and woody structure less than 2 m tall increased. The bobwhite also occupies cottonwood reforestation plots less than 4 years old in Mississippi and Louisiana (Twedt and others 2002). Most management for this species has been at the local scale, but Guthery (1999) showed that optimal configuration of patch types and sizes has variability (slack), and Williams and others (2004) promoted a regional management strategy that focused on useable space (i.e., more patches of native prairies, savanna, and other favored vegetation types).

Weather affects bobwhite populations, including positive effects of summer temperature and fall precipitation (Lusk and others 2002a) and negative effects of spring flooding and low winter temperatures (Applegate and others 2002). Bridges and others (2001) found a negative correlation between drought indices in dry regions and bobwhite abundance, but this pattern did not hold in wetter regions of Texas. Lusk and others (2002b) also found that climatic variables were more important than landscape variables for predicting bobwhite abundance in Oklahoma.

Nests are constructed of litter (grass or pine needles) in areas of high structural complexity (Townsend and others 2001); brood cover is found in open areas with dense forbs that still permit mobility at ground level. Nevertheless, Taylor and others (1999) did not find any habitat attributes associated with higher probabilities of adult survival or nest success. White and others (2005) examined multiple landscape buffers (radii of 250 to 1,000 m) around nest sites and random points to examine landscape effects on nest site selection. Bobwhite responded to both composition and configuration of landscapes, including proportions of open-canopy planted pine and fallow fields, interspersion-juxtaposition index, and patch density. A model containing all four of these variables applied at the largest landscape had the best predictive ability, but was closely followed by a model containing only proportion of open-canopy planted pine applied at the smallest landscape size. Several other types of habitat models have been developed for the bobwhite: HSI (Schroeder 1985), PATREC (Roseberry and Sudkamp 1998), and logistic regression (Burger and others 2004). Tests of these models showed that they perform poorly (Roseberry and Sudkamp 1998, Burger and others 2004, Jones-Farrand and Millspaugh 2006).

## Model Description

Habitat quality for bobwhite is affected by many parameters that are not measured easily at any scale: the proportion of forbs or open areas in grasslands, herbaceous vegetation height, grasslands and crop-field management, and intra- and inter-annual climatic variations. Therefore, we restricted our habitat suitability model to aspects of landscape composition and forest structure that were quantifiable from available datasets. Our final model includes seven variables: landform, landcover, successional age class, hardwood basal area, evergreen basal area, grass landcover, and interspersion of open and forest habitats.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 87). We directly assigned SI scores to these combinations on the basis of habitat associations for the northern bobwhite outlined in Hamel (1992).

Forested sites used by the northern bobwhite typically are woodlands with low hardwood and pine basal area (SI2 and SI3, respectively). We used data from Cram and others (2002) and Palmer and Wellendorf (2006) to inform inverse logistic functions that predict SI scores for the bobwhite at various basal area levels (Tables 88-89; Figs 51-52).

We directly assigned SI scores to grass landcover (SI4) classes based on their potential to provide feeding, nesting, and brood-rearing habitat (Guthery 1997) (Table 90). We assumed that natural grassland-herbaceous landcovers had the greatest potential to provide these habitats, though it is likely that a given patch can satisfy only two of the three requisites

**Table 87.—Relationship of landform, landcover type, and successional age class to suitability index (SI) scores for northern bobwhite habitat; values in parentheses apply to West Gulf Coastal Plain/Ouachitas**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.167 | 0.167 | 0.083 | 0.000 | 0.000 |
| | Deciduous | 0.167 | 0.167 | 0.083 | 0.000 | 0.000 |
| | Evergreen | 1.000 | 1.000 | 0.667 | 0.500 | 0.667 |
| | Mixed | 0.667 | 1.000 | 0.667 | 0.333 | 0.333 |
| | Orchard-vineyard | 0.167 | 0.167 | 0.083 | 0.000 | 0.000 |
| | Woody wetlands | 0.334 | 0.334 | 0.250 | 0.250 | 0.334 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.667<br>(1.000) | 1.000 | 0.667 | 0.333<br>(0.500) | 0.333<br>(0.667) |
| | Deciduous | 0.333 | 0.667 | 0.333 | 0.000 | 0.000 |
| | Evergreen | 1.000 | 1.000 | 0.667 | 0.500 | 0.667 |
| | Mixed | 0.667 | 1.000 | 0.667 | 0.333 | 0.333 |
| | Orchard-vineyard | 0.333 | 0.667 | 0.333 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.667<br>(0.834) | 1.000<br>(0.834) | 0.667 | 0.333<br>(0.667) | 0.333<br>(0.667) |
| | Deciduous | 0.333 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Evergreen | 1.000<br>(0.834) | 1.000<br>(0.834) | 0.667 | 0.500<br>(0.667) | 0.667 |
| | Mixed | 0.667 | 1.000 | 0.667 | 0.333 | 0.333 |
| | Orchard-vineyard | 0.333 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |

at any point in time (Stoddard 1931). We assumed that areas in small grain production provided foraging opportunities but had little residual value for nesting or brood rearing. Similarly, fallow fields provide marginal nest and brood habitat but little forage. Finally, pasture-hay and row crops may provide foraging, nesting, and brood-rearing habitat but their value likely is limited due to management practices that produce unsuitable vegetative structure during most of the breeding season.

The bobwhite relies on landscapes comprised of interspersed vegetation types (White and others 2005, Guthery 2000). We used the composition of open and forest landcovers within a 1-km landscape (SI5) to index the interspersion of these cover types. Guthery (1999, 2000) and others before him (see Schroeder 1985 and references therein) have noted that this species can tolerate a broad range of landscape configurations. On the basis of suggestions from Fred Guthery (2006, Oklahoma State University, pers. commun.), we assumed that high quality habitat was characterized by 10 to 40 percent forest land and 60 to 90 percent open habitat (Table 91).

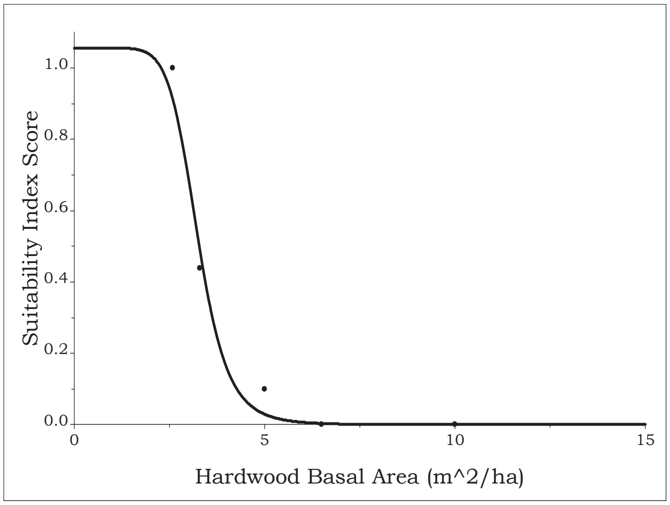


Figure 51.—Relationship between hardwood basal area and suitability index (SI) scores for northern bobwhite habitat. Equation: SI score = 1/ (1.000 + (0.053 * (hardwood basal area)$^{5.068}$)).

**Table 88.—Influence of hardwood basal area on suitability index (SI) scores for northern bobwhite habitat**

| Hardwood basal area (m$^2$/ha) | SI score |
|---|---|
| 0.0[a] | 1.000 |
| 2.6[b] | 1.000 |
| 3.3[b] | 0.439 |
| 5.0[a] | 0.100 |
| 6.5[b] | 0.000 |
| 10.0[a] | 0.000 |

[a]Assumed value.
[b]Cram and others (2002).

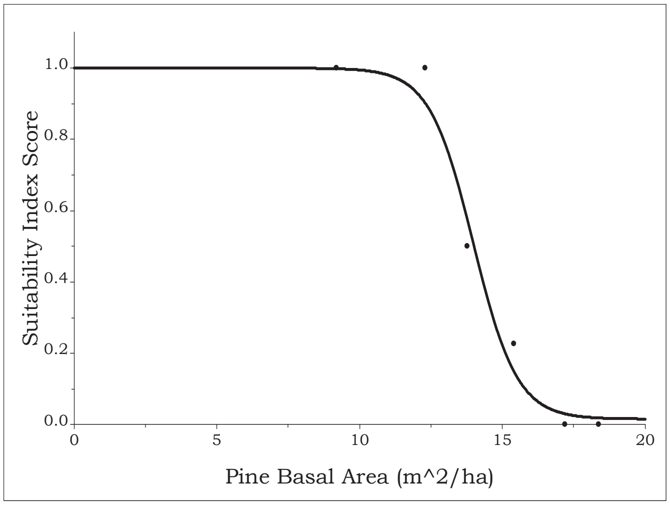


Figure 52.—Relationship between pine basal area and suitability index (SI) scores for northern bobwhite habitat. Equation: SI score = 1 - (0.984 / (1 + (83605490 * e$^{-1.305 * \text{pine basal area}}$))).

**Table 89.—Influence of pine basal area on suitability index (SI) scores for northern bobwhite habitat**

| Pine basal area (m$^2$/ha) | SI score |
|---|---|
| 0.00[a] | 1.000 |
| 9.20[b] | 1.000 |
| 12.30[a] | 1.000 |
| 13.78[b] | 0.500 |
| 15.40[c] | 0.228 |
| 17.20[c] | 0.000 |
| 18.37[b] | 0.000 |

[a]Assumed value.
[b]Palmer and Wellendorf (2006).
[c]Cram and others (2002).

We calculated the overall HSI score by first determining the geometric mean of SI scores for forest structure attributes (SI1, SI2, and SI3). Open habitats lacking forest structure were assigned SI score independently (SI4). The landscape context of these forest and open habitats were incorporated into the HSI calculation by determining the geometric mean of these site-level and landscape-level variables (SI5) together.

Overall HSI = $(((SI1 * SI2 * SI3)^{0.333} + SI4) * SI5)^{0.500}$

**Table 90.—Relationship between open and grassy landcover and suitability index (SI) scores for northern bobwhite habitat**

| Landcover type[a] | SI score |
|---|---|
| Grassland-herbaceous | 1.0 |
| Pasture-hay | 0.1 |
| Row crops | 0.1 |
| Small grains | 0.4 |
| Fallow | 0.2 |

[a] Assumed value.

**Table 91.—Suitability index scores for northern bobwhite habitat based on the proportion of cells providing: 1) good nesting, feeding, and brood-rearing habitat (open landcovers); 2) escape and thermal cover (forest landcovers) within 1-km radius**

| Proportion | Proportion forest[a] | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| open[b] | 0.0 | 0.1 | 0.2 | 0.3 | 0.4 | 0.5 | 0.6 | 0.7 | 0.8 | 0.9 | 1.0 |
| 0.0 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| 0.1 | 0.00 | 0.00 | 0.10 | 0.15 | 0.25 | 0.25 | 0.25 | 0.20 | 0.15 | 0.10 | |
| 0.2 | 0.00 | 0.10 | 0.15 | 0.25 | 0.35 | 0.35 | 0.30 | 0.25 | 0.20 | | |
| 0.3 | 0.00 | 0.30 | 0.35 | 0.45 | 0.45 | 0.45 | 0.40 | 0.30 | | | |
| 0.4 | 0.00 | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | | | | |
| 0.5 | 0.00 | 0.70 | 0.70 | 0.70 | 0.70 | 0.70 | | | | | |
| 0.6 | 0.00 | 0.90 | 0.90 | 0.90 | 0.90 | | | | | | |
| 0.7 | 0.00 | 0.90 | 1.00 | 1.00 | | | | | | | |
| 0.8 | 0.00 | 0.90 | 1.00 | | | | | | | | |
| 0.9 | 0.00 | 0.90 | | | | | | | | | |
| 1.0 | 0.00 | | | | | | | | | | |

[a] Forest = landcovers with positive SI1 score (Table 87).

[b] Open = landcovers identified in SI4 (Table 90).

## Verification and Validation

The northern bobwhite was found in all 88 subsections of the CH and WGCP. Spearman rank correlation support a significant ($P = 0.006$) positive association ($r_s = 0.29$) between average HSI score and mean BBS route abundance across subsections. The generalized linear model predicting BBS abundance from BCR and HSI for the northern bobwhite was significant ($P \leq 0.001$; $R^2 = 0.440$); however, the coefficient on the HSI predictor variable was negative ($\beta = -37.119$). Therefore, we considered the HSI model for the northern bobwhite verified but not validated (Tirpak and others 2009a).

## Northern Parula

Chandler S. Robbins, Patuxent Bird Identification InfoCenter
Photo used with permission

### Status

The northern parula (*Parula americana*), a long-distance neotropical migrant, breeds in two disjunct zones of eastern North America: New England-southern Canada and the southeastern United States. This species is notably absent from the southern Great Lakes. It depends on epiphytes—Spanish moss in the south and old man's beard in the north—as a nesting substrate. Parula populations have been stable in most regions during the last 40 years and have increased in some areas including the CH (Table 5). This species is not considered a Bird of Conservation Concern in the CH or WGCP (regional combined score = 12 and 13, respectively; Table 1).

### Natural History

The northern parula is common in the bottomland hardwood and riverine forests of the Southeastern United States (Moldenhauer and Regelski 1996). It also occupies mixed pine-hardwoods, though at lower densities (Moldenhauer and Regelski 1996). The northern parula has two competing habitat requirements: a preference for canopy gaps and large forest blocks. Moorman and Guynn (2001) found that this species is more abundant near canopy gaps than forest-interior sites with an unbroken canopy in bottomland hardwoods, and Annand and Thompson (1997) observed the highest northern parula densities in forests with canopy gaps resulting from single-tree selection. However, the probability of detecting the northern parula increases with riparian buffer width (Kilgo and others 1998) and forest patch size (Robbins and others 1989).

The northern parula forages in the mid- to upper canopy layers (Moldenhauer and Regelski 1996), so it is not surprising that it prefers microsites with high basal area (Robbins and others 1989), high canopy cover, and tall canopies (James 1971), and avoids areas with dense understories (often associated with open canopies) (Torres and Leberg 1996). In the Southeast, this species nests almost exclusively in Spanish moss (Moldenhauer and Regelski 1996). However, no studies have identified Spanish moss as limiting.

### Model Description

The HSI model for the northern parula includes six variables: landform, landcover, successional age class, forest patch size, percent forest in a 1-km radius, and canopy cover.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 92). We directly assigned SI scores to these combinations on the basis of habitat associations of northern parulas reported by Hamel (1992) for the Southeast.

We derived a logarithmic function (Fig. 53) from data on the occupancy rate of northern parulas in forest blocks of varying size (SI2; Hayden and others 1985, Robbins and others 1989) (Table 93) to predict habitat suitability from patch area. However, small forest

**Table 92.—Relationship of landform, landcover type, and successional age class to suitability index scores for northern parula habitat**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.083 | 0.500 | 0.834 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.250 | 0.750 | 1.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.250 | 0.500 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.167 | 0.667 | 1.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.167 | 0.333 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.167 | 0.667 | 1.000 |

patches in predominantly forested landscapes may provide habitat due to their proximity to large forest blocks (Rosenberg and others 1999). To capture this relationship, we fit a logistic function (Fig. 54) to data (Table 94) derived from Donovan and others (1997), who observed differences in predator and brood parasite communities among highly fragmented (< 15 percent), moderately fragmented (45 to 50 percent), and lightly fragmented (> 90 percent forest) landscapes. We assumed that the midpoints between these classes (30 and 70 percent forest) defined the specific cutoffs for poor (SI score ≤ 0.10) and excellent (SI score ≥ 0.90) habitat, respectively. We used the maximum SI score from SI2 or SI3 to account for small patches in predominantly forested landscapes.

We included canopy cover (SI4) in our model to capture the preference of the northern parula for interior edges. James (1971), Collins and others (1982), and Morgan and Freedman (1986) found that the northern parula is associated with increased canopy cover. Nonetheless, there seems to be a threshold above which suitability declines. Robbins and others (1989) observed an inverse relationship between canopy cover and northern parula abundance, and Annand and Thompson (1997) observed a threefold increase of parulas in single-tree selection stands characterized by a heterogeneous canopy than in mature forest habitats with closed canopies. On the basis of these studies, we assumed that

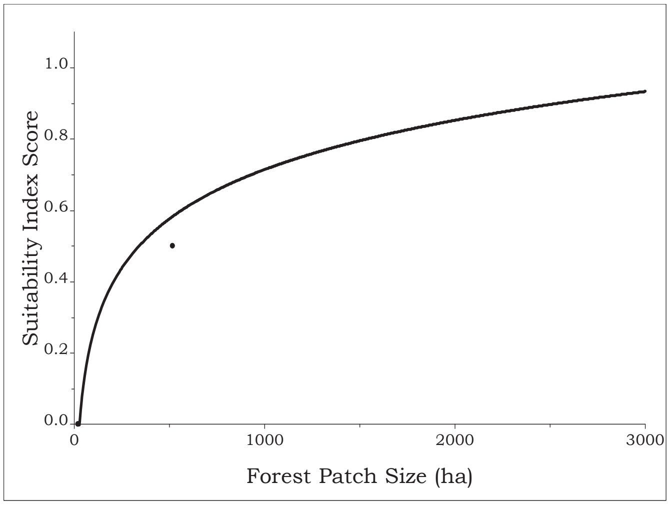


Figure 53.—Relationship between forest patch size and suitability index (SI) scores for northern parula habitat. Equation: SI score = 0.199 * ln(forest patch size) – 0.661.

**Table 93.—Influence of forest patch size on suitability index (SI) scores for northern parula habitat**

| Forest patch size (ha) | SI score |
|---|---|
| 23.6[a] | 0.0 |
| 520[b] | 0.5 |
| 3,200[b] | 1.0 |

[a] Hayden and others (1985).

[b] Robbins and others (1989).

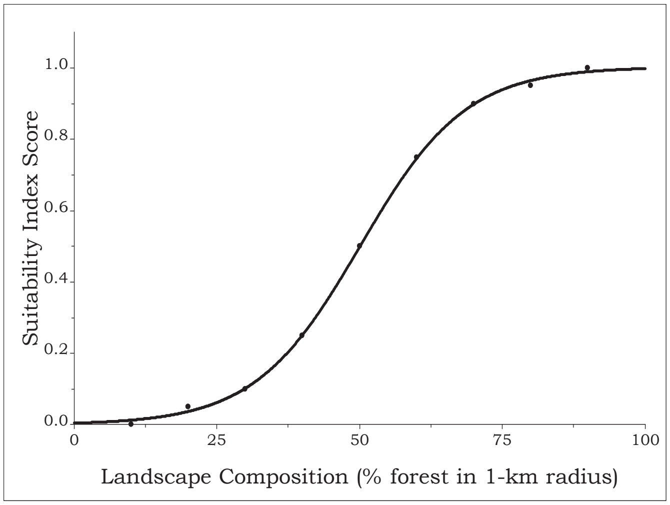


Figure 54.—Relationship between local landscape composition and suitability index (SI) scores for northern parula habitat. Equation: SI score = 1.005 / (1.000 + (221.816 * $e^{-0.108 * (\text{local landscape composition})}$)).

**Table 94.—Relationship between local landscape composition (percent forest in 1-km radius) and suitability index (SI) scores for northern parula habitat**

| Landscape composition | SI score |
|---|---|
| 0[a] | 0.00 |
| 10[a] | 0.00 |
| 20[a] | 0.05 |
| 30[b] | 0.10 |
| 40[a] | 0.25 |
| 50[b] | 0.50 |
| 60[a] | 0.75 |
| 70[b] | 0.90 |
| 80[a] | 0.95 |
| 90[b] | 1.00 |
| 100[a] | 1.00 |

[a] Assumed value.

[b] Donovan and others (1997).

habitat suitability was optimal at 90 percent canopy cover and decreased as the canopy became increasingly open or closed. We fit an inverse quadratic function (Fig. 55) to data demonstrating this relationship (Table 95).

To calculate the overall HSI score, we determined the geometric mean of SI scores for forest structure attributes (SI1 and SI4) and then calculated the geometric mean of this value and landscape composition (Max of SI2 or SI3).

Overall HSI = $((SI1 * SI4)^{0.500} * \text{Max}(SI2 \text{ or } SI3))^{0.500}$

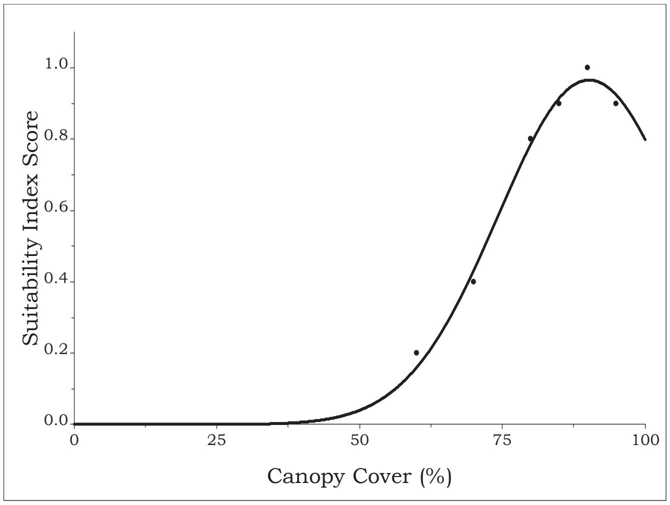


Figure 55.—Relationship between canopy cover and suitability index (SI) scores for northern parula habitat. Equation: SI score = 1 / (37.3645 – (0.8127 * canopy cover) + (0.00454 * (canopy cover$^2$))).

**Table 95.—Influence of canopy cover on suitability index (SI) scores for northern parula habitat**

| Canopy cover (percent)[a] | SI score |
|---|---|
| 60 | 0.2 |
| 70 | 0.4 |
| 80 | 0.8 |
| 85 | 0.9 |
| 90 | 1.0 |
| 95 | 0.9 |
| 100 | 0.8 |

[a] Assumed value.

## Verification and Validation

The northern parula was found in all 88 subsections of the CH and WGCP. Spearman rank correlation identified a significant ($P \leq 0.001$) positive relationship ($r_s = 0.51$) between average HSI score and mean BBS route abundance across subsections. The generalized linear model predicting BBS abundance from BCR and HSI for the northern parula was significant ($P \leq 0.001$; $R^2 = 0.276$), and the coefficient on the HSI predictor variable was both positive ($\beta = 5.250$) and significantly different from zero ($P \leq 0.001$). Therefore, we considered the HSI model for the northern parula both verified and validated (Tirpak and others 2009a).

## Orchard Oriole

Deanna K. Dawson,
Patuxent Bird Identification InfoCenter
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### Status

The orchard oriole (*Icterus spurius*), a neotropical migrant, is found throughout most of the United States east of the Rocky Mountains except for New England and the northern Great Lakes. Although this species has experienced increases along the edges of its distribution, populations have declined in the core of its range where densities are highest. In the WGCP, populations have declined by 3 percent per year since 1967 (Table 5). Populations in the adjacent Mississippi Alluvial Valley have declined 4 percent. The orchard oriole is a Bird of Conservation Concern in the WGCP and has been identified as a species requiring management attention in both the CH and WGCP (regional combined score = 17 and 18, respectively; Table 1).

### Natural History

The orchard oriole breeds in wooded riparian zones, floodplains, marshes, and shorelines (Scharf and Kren 1996) but also in open shrublands and low-density human-dominated areas (e.g., farms and parklands). It is semi-colonial in optimal habitat but relatively solitary in marginal areas. This species is a common host of the brown-headed cowbird.

### Model Description

The HSI model for the orchard oriole includes five variables: landform, landcover, successional age class, forest within a 1-km radius, and basal area.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 96). We directly assigned SI scores to these combinations based on vegetation and successional age class associations in Hamel (1992). However, we adjusted Hamel's values to account for the preference of the orchard oriole for mesic habitats (e.g., riparian zones, floodplains, and marshes; Scharf and Kren 1996).

The orchard oriole is not area sensitive but generally is restricted to forested landscapes. Therefore, we included only local forest composition (SI2) in our model to discount forest patches that were isolated within a matrix of nonforest landcover. Conversely, this is an edge species whose abundance declines in heavily forested regions (Scharf and Kren 1996). Therefore, we assumed that landscapes with 70 to 80 percent forest provided optimal habitat suitability and reduced suitability symmetrically as landscape composition shifted from these optima (Table 97, Fig. 56).

This species is most abundant in areas with scattered trees. Heltzel and Leberg (2006) observed significantly fewer orioles in stands with an average basal area of 25 $m^2$ per ha than in recently harvested stands with an average basal area of 18 $m^2$ per ha. We assumed that habitat suitability was optimal for the orchard oriole at lower basal areas and modeled

**Table 96.—Relationship of landform, landcover type, and successional age class to suitability index scores for orchard oriole habitat**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.500 | 1.000 | 1.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.500 | 1.000 | 1.000 |
| | Deciduous | 0.000 | 0.000 | 0.500 | 1.000 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.500 | 1.000 | 1.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.500 | 1.000 | 1.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.333 | 0.667 | 0.667 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.250 | 0.500 | 0.500 |
| | Deciduous | 0.000 | 0.000 | 0.250 | 0.500 | 0.500 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.333 | 0.667 | 0.667 |
| | Woody wetlands | 0.000 | 0.000 | 0.500 | 1.000 | 1.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.333 | 0.667 | 0.667 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.250 | 0.500 | 0.500 |
| | Deciduous | 0.000 | 0.000 | 0.250 | 0.500 | 0.500 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.333 | 0.667 | 0.667 |
| | Woody wetlands | 0.000 | 0.000 | 0.500 | 1.000 | 1.000 |

the basal area (SI3)-habitat suitability relationship as a quadratic function (Fig. 57) that maximized SI scores at intermediate basal area values (12.5 $m^2$/ha; Table 98).

To calculate the overall HSI score, we determined the geometric mean of SI scores for forest structure indices (SI1 and SI3) and then determined the geometric mean of this value and landscape composition (SI2).

Overall HSI = $((SI1 * SI3)^{0.500} * SI2)^{0.500}$

## Verification and Validation

The orchard oriole was found in all 88 subsections of the CH and WGCP. Spearman rank correlation identified a significant ($P \leq 0.001$) positive relationship ($r_s = 0.34$) between average HSI score and mean BBS route abundance across subsections. The generalized linear model predicting BBS abundance from BCR and HSI for the orchard oriole was significant ($P = 0.088$; $R^2 = 0.056$), and the coefficient on the HSI predictor variable was positive ($\beta = 2.442$) but not significantly different from zero ($P = 0.221$). Therefore, we considered the HSI model for the orchard oriole verified but not validated (Tirpak and others 2009a).

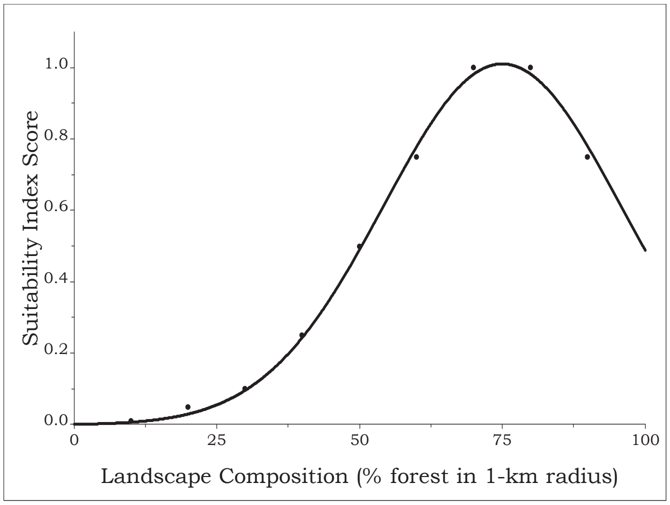

Figure 56.—Relationship between landscape composition and suitability index (SI) scores for orchard oriole habitat. Equation: SI score = $1.011 * e^{((0 - ((\text{landscape composition} * 100) - 74.945) \,\hat{}\, 2) / 863.949)}$.

**Table 97.—Relationship between landscape composition (percent forest in 1-km radius) and suitability index (SI) scores for orchard oriole habitat**

| Landscape composition[a] | SI score |
|---|---|
| 0 | 0.00 |
| 10 | 0.00 |
| 20 | 0.05 |
| 30 | 0.10 |
| 40 | 0.25 |
| 50 | 0.50 |
| 60 | 0.75 |
| 70 | 1.00 |
| 80 | 1.00 |
| 90 | 0.75 |
| 100 | 0.50 |

[a] Assumed value.

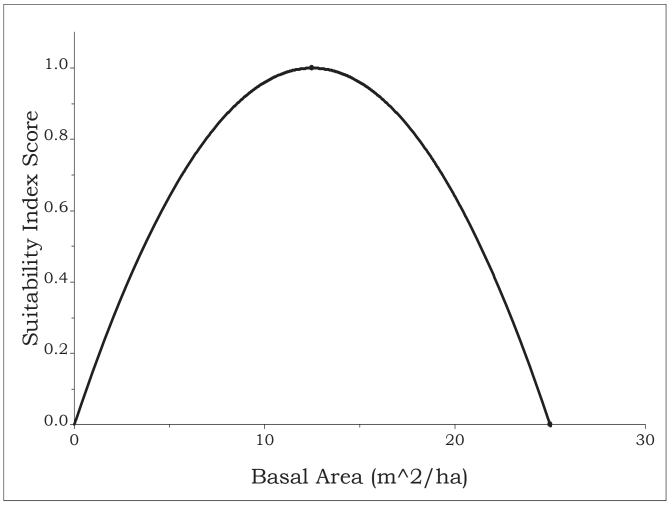

Figure 57.—Relationship between basal area and suitability index (SI) scores for orchard oriole habitat. Equation:
SI score = (0.16 * basal area) - (0.00639 * (basal area$^2$)).

**Table 98.—Influence of basal area (m$^2$/ha) on suitability index (SI) scores for orchard oriole habitat**

| Basal area (m$^2$/ha) | SI score |
|---|---|
| 0.0[a] | 0.0 |
| 12.5[a] | 1.0 |
| 25.0[b] | 0.0 |

[a] Assumed value.
[b] Heltzel and Leberg (2006).

## Painted Bunting


Deanna K. Dawson,
Patuxent Bird Identification InfoCenter
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### Status

The painted bunting (*Passerina cyanea*) occurs as two allopatric populations that may represent separate species (Lowther and others 1999). The western population inhabits the southern Great Plains and the western edges of the CH and WGCP, while the eastern population inhabits the Atlantic Coastal Plain from North Carolina to Florida. Populations have been relatively stable across the WGCP as a whole (Table 5), but populations have declined in Arkansas (5.8 percent per year from 1967 to 2004), Louisiana (3.5 percent), and Texas (2.4 percent) but increased in Oklahoma (1.3 percent; Sauer and others 2005). The painted bunting is not an FWS Bird of Conservation Concern but is a PIF management attention priority in both the CH and WGCP (regional combined score = 16 and 17, respectively; Table 1).

### Natural History

The habitat requirements of the painted bunting are poorly understood. This species generally occupies areas of scattered woody vegetation. Kopachena and Crist (2000a) characterized painted bunting habitat in northeast Texas as "wooded areas in otherwise open habitat" as opposed to the indigo bunting, which occurs in "open areas in otherwise wooded habitat." The painted bunting use smaller, more heterogeneous groups of trees than the indigo bunting, but microhabitats differ little between these species (Kopachena and Crist 2000b). The painted bunting occupies narrow riparian strips in eastern Texas and its abundance decreases quickly as widths exceed 70 m (Conner and others 2004).

The painted bunting nests in low, woody vegetation (Lowther and others 1999) and its territory size varies with its population density. In Missouri, territories ranged from 0.64 to 6.66 ha and included 80 percent pasture and 20 percent woodland. This species is a common host of both the brown-headed and bronzed cowbird.

### Model Description

The HSI model for the painted bunting includes six variables: landform, landcover, successional age class, distance to edge, interspersion of open and forested lands, and small stem (< 2.5 cm d.b.h.) density.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 99). We directly assigned SI scores to these combinations on the basis of relative habitat rankings for vegetation and successional age class associations of painted buntings reported by Hamel (1992). We assigned higher values to the shrub-seedling age class than Hamel (1992) on the basis of qualitative descriptions in Lowther and others (1999).

An early-successional species, the painted bunting is associated with edges. We used data on territory density from Lanyon and Thompson (1986; Table 100) to define an inverse logistic function linking SI scores to distance from an edge (SI2; Fig. 58).

**Table 99.—Relationship of landform, landcover type, and successional age class to suitability index scores for painted bunting habitat**

| Landform | Landcover type | Successional age class | | | | |
|---|---|---|---|---|---|---|
| | | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.500 | 0.500 | 0.250 | 0.000 |
| | Deciduous | 0.000 | 0.500 | 0.500 | 0.250 | 0.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.500 | 0.500 | 0.250 | 0.000 |
| | Woody wetlands | 0.000 | 1.000 | 0.750 | 0.500 | 0.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.500 | 0.500 | 0.250 | 0.000 |
| | Deciduous | 0.000 | 0.500 | 0.500 | 0.250 | 0.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.500 | 0.500 | 0.250 | 0.000 |
| | Woody wetlands | 0.000 | 1.000 | 0.750 | 0.500 | 0.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.500 | 0.500 | 0.250 | 0.000 |
| | Deciduous | 0.000 | 0.500 | 0.500 | 0.250 | 0.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.500 | 0.500 | 0.250 | 0.000 |
| | Woody wetlands | 0.000 | 1.000 | 0.750 | 0.500 | 0.000 |

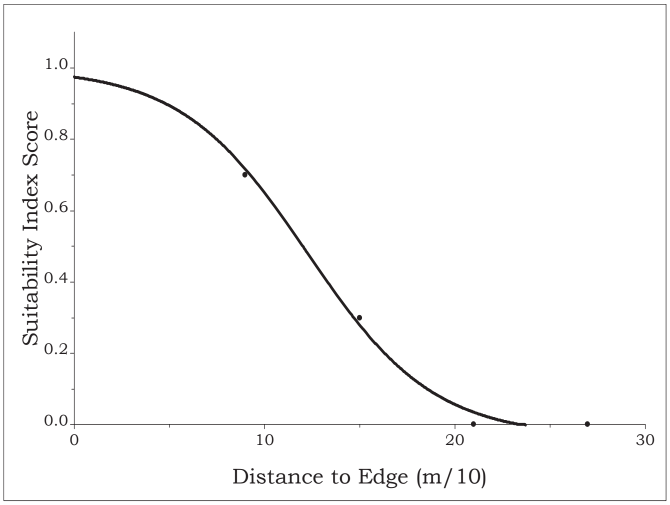


Figure 58.—Relationship between distance to edge and suitability index (SI) scores for painted bunting habitat. Equation: SI score = 1 - (1.034 / (1 + (39.685 * $e^{-0.301 * (\text{distance to edge} / 10\text{ m})}$))).

**Table 100.—Influence of distance to edge on suitability index (SI) scores for painted bunting habitat**

| Distance to edge (m) | SI score |
|---|---|
| 0[a] | 1.0 |
| 90[a] | 0.7 |
| 150[a] | 0.3 |
| 210[a] | 0.0 |
| 270[b] | 0.0 |

[a] Lanyon and Thompson (1986).
[b] Assumed value.

**Table 101.—Suitability index scores for painted bunting habitat based on the proportion of open and forest landcovers within 5-ha area**

| Proportion forest[b] | Proportion open[a] 0.0 | 0.1 | 0.2 | 0.3 | 0.4 | 0.5 | 0.6 | 0.7 | 0.8 | 0.9 | 1.0 |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| 0.1 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | |
| 0.2 | 0.0 | 0.0 | 0.5 | 0.5 | 0.5 | 0.5 | 0.5 | 0.5 | 0.5 | | |
| 0.3 | 0.0 | 0.0 | 0.5 | 0.7 | 0.7 | 0.7 | 0.7 | 0.7 | | | |
| 0.4 | 0.0 | 0.0 | 0.5 | 0.7 | 0.9 | 0.9 | 0.9 | | | | |
| 0.5 | 0.0 | 0.0 | 0.5 | 0.7 | 0.9 | 1.0[c] | | | | | |
| 0.6 | 0.0 | 0.0 | 0.5 | 0.7 | 0.9 | | | | | | |
| 0.7 | 0.0 | 0.0 | 0.5 | 0.7 | | | | | | | |
| 0.8 | 0.0 | 0.0 | 0.5 | | | | | | | | |
| 0.9 | 0.0 | 0.0 | | | | | | | | | |
| 1.0 | 0.0 | | | | | | | | | | |

[a] Open = herbaceous natural, cultivated, and emergent herbaceous wetland.

[b] Forest = upland forested, transitional, woody wetland, and orchard/vineyard.

[c] Unpublished data.

The presence of both forest and open landcovers in the landscape (SI3) is perhaps the most important component of painted bunting habitat. We maximized SI scores for this species in landscapes containing 50 percent forest and 50 percent open habitats based on unpublished data (Jeffrey Kopachena, 2006, Texas A&M University—Commerce, pers. commun.). Norris and Elder (1982, cited in Lowther and others 1999) observed the painted bunting in landscapes with forest cover of 20 to 80 percent forest. We used these values as cutoffs for forest cover in our interspersion function for the painted bunting (Table 101).

As an early successional species, the painted bunting occupies habitats containing high densities of small stems (SI4). We assumed that the mean stem density values (6,400 stems/ha) reported by Kopachena and Crist (2000b) were characteristic of average habitat suitability (SI score = 0.500). However, because of the high standard error (6,300 stems/ha) associated with this estimate, we assumed that a stem density that was twice the mean was necessary to ensure optimal habitat (Table 102). We fit a smoothed logistic function through these data points (Fig. 59) to quantify the relationship between small stem density and SI scores for painted bunting habitat.

To calculate the HSI score for sapling and pole successional age class stands, we determined the geometric mean of SI scores for forest structure (SI1 and SI4) and landscape composition (SI2 and SI3) separately and then the geometric mean of these means together.

$$HSI_{Sap\text{-}pole} = ((SI1 * SI4)^{0.500} * (SI2 * SI3)^{0.500})^{0.500}$$

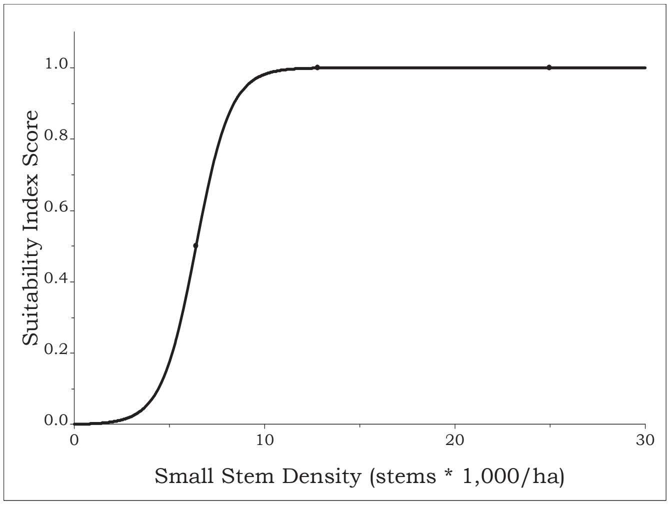


Figure 59.—Relationship between small stem (< 2.5 cm d.b.h.) density (stems * 1000/ha) and suitability index (SI) scores for painted bunting habitat. Equation: SI score = (1.000 / (1 + (1178.674 * e $^{-1.105 * (\text{small stem density} / 1000)}$))).

**Table 102.—Influence of small stem density (stems * 1,000/ha) on suitability index (SI) scores for painted bunting habitat**

| Small stem density | SI score |
|---|---|
| 0.0[a] | 0.0 |
| 6.4[b] | 0.5 |
| 12.8[a] | 1.0 |
| 25.0[a] | 1.0 |

[a] Assumed value.

[b] Kopachena and Crist (2000b).

We assumed that shrub-seedling successional age class stands were suitable regardless of edge or landscape composition. Thus, we calculated the HSI score as the geometric mean of forest structure attributes alone (SI1 and SI4).

$$HSI_{Shrub} = (SI1 * SI4)^{0.500}$$

The overall HSI score is the sum of the two age class specific SIs:

$$\text{Overall HSI} = SI_{Sap\text{-}pole} + SI_{Shrub}$$

## Verification and Validation

The painted bunting was found in only 38 of the 88 subsections within the CH and WGCP. Nevertheless, Spearman rank correlations based on either all subsections or only subsections in which the painted bunting occurred produced similar results: significant ($P \leq 0.001$ in both analyses) positive associations ($r_s = 0.56$ and 0.58, respectively) between average HSI score and mean BBS route abundance at the subsection scale. The generalized linear model predicting BBS abundance from BCR and HSI for the painted bunting was significant ($P \leq 0.001$; $R^2 = 0.480$), and the coefficient on the HSI predictor variable was both positive ($\beta = 70.737$) and significantly different from zero ($P \leq 0.001$). Therefore, we considered the HSI model for the painted bunting both verified and validated (Tirpak and others 2009a).

# Pileated Woodpecker

U.S Forest Service

## Status

The pileated woodpecker (*Dryocopus pileatus*) breeds throughout eastern North America, southern Canada, and the montane forests of the West. Populations have been stable across most of its range, including the WGCP, over the last 40 years and have increased along the northern limit of this bird's distribution. In the CH, populations have increased by 1.8 percent per year since 1967 (Sauer and others 2005) (Table 5). This species is a management attention priority in the WGCP (regional combined score = 16) but has no special conservation status in the CH (regional combined score = 13; Table 1).

## Natural History

The pileated woodpecker uses a variety of forest types across its range but typically is associated with older successional age classes (Bull and Jackson 1995, Annand and Thompson 1997). The key component to pileated woodpecker habitat is an abundance of large snags—the more the better. Different researchers define "large" differently (Renken and Wiggers 1989, Savignac and others 2000, Showalter and Whitmore 2002) but the pileated woodpecker is invariably associated with the largest available size class. In Missouri, this species is associated with bottomland hardwood forest (Renken and Wiggers 1993); in east Texas, the pileated woodpecker is equally abundant in bottomland hardwoods, longleaf pine savanna, and mixed pine-hardwood stands, so long as suitable snags are available (Shackelford and Conner 1997). Closed canopies (canopy cover of 75 to 96 percent) are the norm (Renken and Wiggers 1989). Because it has a large home range (53 to 160 ha), it is not surprising that the pileated woodpecker is sensitive to forest area. Robbins and others (1989) did not detect this species in woodlots less than 42 ha and larger areas likely are required for breeding pairs. Schroeder (1982) considered 130 ha as the minimum forest patch size for this species.

## Model Description

The pileated woodpecker model includes six variables: landform, land cover, successional age class, large snag (> 30 cm d.b.h.) density, forest patch size, and percentage of forest in a 1-km radius.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 103). We used the habitat associations of the pileated woodpecker outlined in Hamel (1992) to assign SI scores to these combinations.

Large snags (SI2) are used for roosting, nesting, and foraging and are an important component of pileated woodpecker habitat. We fit a logistic function (Fig. 60) to data from Renken and Wiggers (1989) on the relative density of this species on sites with varying large snag densities to predict SI scores based on this habitat feature (Table 104).

**Table 103.—Relationship of landform, landcover type, and successional age class to suitability index scores for pileated woodpecker habitat; values in parentheses apply to West Gulf Coastal Plain/Ouachitas**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.042 | 0.083 | 0.167 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.083 | 0.583 | 1.000 |
| | Deciduous | 0.000 | 0.000 | 0.083 | 0.583 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.167 | 0.333 | 0.333 |
| | Mixed | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.167 | 0.667 | 1.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.042 | 0.083 | 0.167 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.167 | 0.500<br>(0.333) | 0.667<br>(0.333) |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.500 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.167 | 0.333 | 0.333 |
| | Mixed | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.167 | 0.667 | 1.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.042 | 0.083 | 0.167 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.167<br>(0.083) | 0.500<br>(0.167) | 0.667<br>(0.167) |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.500 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.167<br>(0.083) | 0.333<br>(0.167) | 0.333<br>(0.167) |
| | Mixed | 0.000 | 0.000 | 0.167 | 0.500 | 0.667 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.167 | 0.667 | 1.000 |

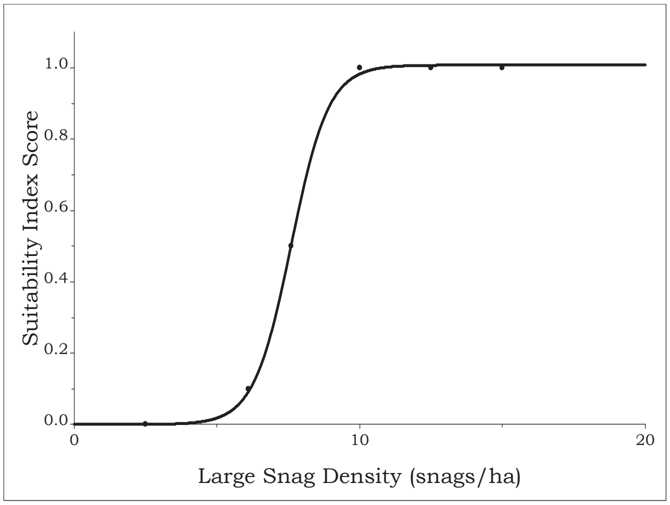


Figure 60.—Relationship between large snag (> 30 cm d.b.h.) density and suitability index (SI) scores for pileated woodpecker habitat. Equation: SI score = (1.0054 / (1 + (747.0936 * $e^{-0.8801 \text{ * large snag density}}$))).

**Table 104.—Influence of large snag (> 30 cm d.b.h.) density (snags/ha) on suitability index (SI) scores for pileated woodpecker habitat**

| Large snag density | SI score |
|---|---|
| 0.[a] | 0.0 |
| 2.5[a] | 0.0 |
| 6.1[b] | 0.1 |
| 7.6[b] | 0.5 |
| 10.0[b] | 1.0 |
| 15.0[a] | 1.0 |
| 12.5[a] | 1.0 |

[a] Assumed value.

[b] Renken and Wiggers (1989).

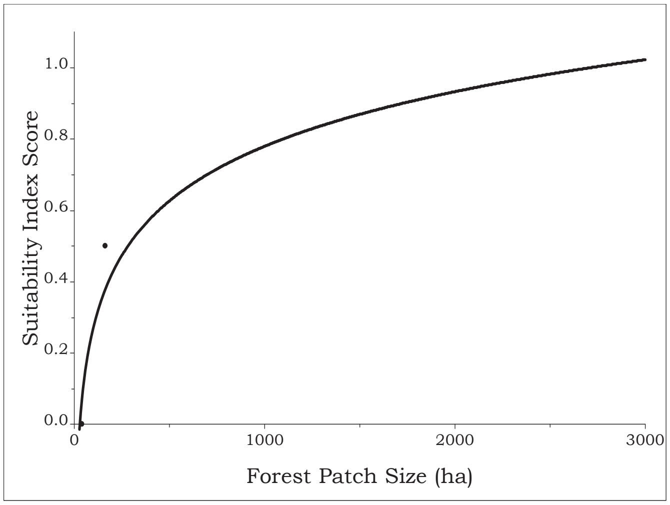


Figure 61.—Relationship between forest patch size and suitability index (SI) scores for pileated woodpecker habitat. Equation: SI score = 0.230 * ln(forest patch size) – 0.877.

**Table 105.—Influence of forest patch size on suitability index (SI) scores for pileated woodpecker habitat**

| Forest patch size (ha)[a] | SI score |
|---|---|
| 42.2 | 0.0 |
| 165 | 0.5 |
| 3,200 | 1.0 |

[a] Robbins and others (1989).

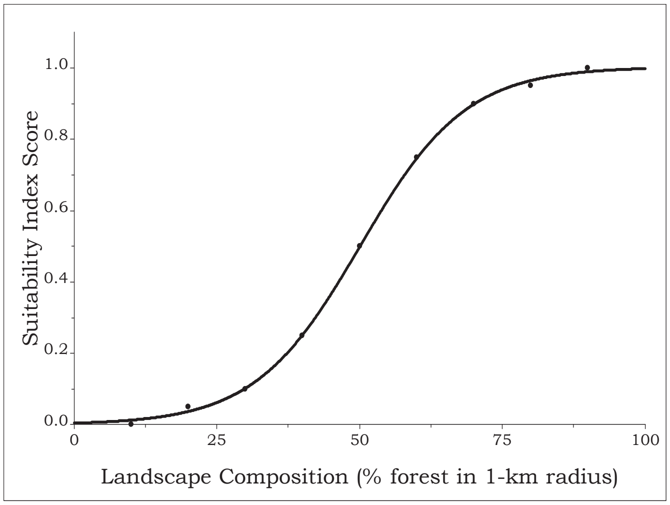


Figure 62.—Relationship between landscape composition and suitability index (SI) scores for pileated woodpecker habitat. Equation: SI score = 1.005 / (1.000 + (221.816 * $e^{-0.108 * \text{(local landscape composition)}}$)).

**Table 106.—Relationship between landscape composition (percent forest in 1-km radius) and suitability index (SI) scores for pileated woodpecker habitat**

| Landscape composition | SI score |
|---|---|
| 0[a] | 0.00 |
| 10[a] | 0.00 |
| 20[a] | 0.05 |
| 30[b] | 0.10 |
| 40[a] | 0.25 |
| 50[b] | 0.50 |
| 60[a] | 0.75 |
| 70[b] | 0.90 |
| 80[a] | 0.95 |
| 90[b] | 1.00 |
| 100[a] | 1.00 |

[a] Assumed value.

[b] Donovan and others (1997).

We incorporated forest patch size (SI3) and percent forest in the local landscape (SI4) as predictors of habitat suitability. Large home ranges for the pileated woodpecker necessitate large forest patches. We fit a logarithmic function (Fig. 61) to data from Robbins and others (1989) on the effect of forest patch size on occupancy rates (Table 105). We also included percent forest in the landscape because small forest patches that may not be used in predominantly nonforested landscapes may provide habitat in predominantly forested landscapes due to their proximity to large forest blocks (Rosenberg and others 1999). To capture this relationship, we fit a logistic function (Fig. 62) to data (Table 106) derived from Donovan and others (1997), who observed differences in predator and brood parasite communities among highly fragmented (< 15 percent), moderately fragmented (45 to 50 percent), and lightly fragmented (> 90 percent forest) landscapes. We assumed that the midpoints between these classes (30 and 70 percent forest) defined the specific cutoffs for poor (SI score ≤ 0.10) and excellent (SI score ≥ 0.90) habitat, respectively. We used the maximum SI score from SI3 or SI4 to account for the higher suitability of small forest patches in predominantly forested landscapes.

To calculate the overall HSI score, we determined the geometric mean of SI scores for forest structure attributes (SI1 and SI2) and multiplied that by the maximum value of forest patch size (SI3) or percent forest in the 1-km radius landscape (SI4) and calculated the geometric mean of that product.

$$\text{Overall HSI} = ((\text{SI1} * \text{SI2})^{0.500} * \text{Max}(\text{SI3 or SI4}))^{0.500}$$

## Verification and Validation

The pileated woodpecker was observed in all 88 subsections of the CH and WGCP. Spearman rank correlation identified a significant ($P \leq 0.002$) positive association ($r_s$ = 0.33) between average HSI score and mean BBS route abundance across subsections. The generalized linear model predicting BBS abundance from BCR and HSI for the pileated woodpecker was significant ($P \leq 0.001$; $R^2 = 0.313$), and the coefficient on the HSI predictor variable was both positive ($\beta = 8.852$) and significantly different from zero ($P \leq 0.001$). Therefore, we considered the HSI model for the pileated woodpecker both verified and validated (Tirpak and others 2009a).

## Prairie Warbler

Deanna K. Dawson,
Patuxent Bird Identification InfoCenter
Photo used with permission

### Status

The prairie warbler (*Dendroica discolor*), a neotropical migrant, occupies early successional habitats throughout the eastern United States. Like many early successional species, populations of this bird have declined throughout the eastern and central United States since 1967, including a drop of 2.6 percent per year in the CH and 4.4 percent per year in the WGCP (Table 5). The prairie warbler is an FWS Bird of Conservation Concern and a management attention priority in both BCRs (regional combined score = 18 in the CH and WGCP; Table 1).

### Natural History

The prairie warbler breeds in shrubby vegetation under an open canopy (Nolan and others 1999). Typical associations in the CH and WGCP include shrubby southern pine forest, pine barrens, scrub oak barrens, abandoned fields and pastures, regenerating forest, abandoned orchards, grassland-forest edge, Christmas tree farms, and reclaimed strip mine spoils. The prairie warbler uses a variety of landforms from xeric uplands in Arkansas to palustrine swamps in Virginia. In comparison to other early successional warblers, this bird occupies sites with fewer dense shrubs than the blue-winged warbler, more dense vegetation and drier areas than the yellow warbler, and less dense vegetation and higher vegetation strata than the common yellowthroat or yellow-breasted chat (Nolan and others 1999).

The prairie warbler nests in shrubs and small trees that are more than 20 m from a field-forest edge (Nolan and others 1999, Woodward and others 2001). However, in eastern Texas this species typically occurs in narrow riparian zones, with abundance decreasing quickly as widths increase (Conner and others 2004). Mean territory size varies inversely with population density, ranging from 0.2 to 3.5 ha in Indiana (Nolan and others 1999). Territory size also varies with shape of forest patch; it is larger in more linear patches. Although males do not limit movements to their defended territory, a female's home range usually is contained within a male's defended territory. This species is a cowbird host. Although parasitism has little effect on hatching success, it can significantly reduce fledging rates.

### Model Description

Our HSI model for the prairie warbler includes seven variables: landform, landcover, successional age class, early-successional patch size, small stem (< 2.5 cm d.b.h.) density, edge occurrence, and canopy cover.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 107). We directly assigned SI scores to these combinations on the basis of habitat associations for the prairie warbler documented in Hamel (1992).

**Table 107.—Relationship of landform, landcover type, and successional age class to suitability index scores for prairie warbler habitat; values in parentheses apply to West Gulf Coastal Plain/Ouachitas**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.333 | 0.167 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.333 | 0.167 | 0.000 | 0.000 |
| | Evergreen | 0.000 | 0.667 | 0.334 | 0.000 | 0.000 |
| | Mixed | 0.000 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.333 | 0.167 | 0.000 | 0.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 1.000<br>(0.667) | 0.500<br>(0.334) | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.667 | 0.333 | 0.000 | 0.000 |
| | Evergreen | 0.000 | 0.667 | 0.334 | 0.000 | 0.000 |
| | Mixed | 0.000 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.333 | 0.167 | 0.000 | 0.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 1.000<br>(0.500) | 0.500<br>(0.250) | 0.000 | 0.000 |
| | Deciduous | 0.000 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Evergreen | 0.000 | 0.667<br>(0.500) | 0.334<br>(0.250) | 0.000 | 0.000 |
| | Mixed | 0.000 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.333 | 0.167 | 0.000 | 0.000 |

Both Woodward and others (2001) and Rodewald and Vitz (2005) observed edge avoidance by this species. Thus, we used a 3 × 3 pixel (90 x 90 m) window to identify early successional habitats (i.e., grass-forb, shrub-seedling, or sapling successional age class forest) adjacent to mature forest stands (i.e., pole or sawtimber successional age class) and reduced the suitability of locations adjacent to edges by half (SI2; Table 108).

We also included early successional patch size (SI3) as an explanatory variable because the prairie warbler is absent from small clearings and edge habitats. We used data from Larson and others (2003) (Table 109) to fit a logistic function (Fig. 63) that characterized the relationship between habitat suitability and early successional patch size.

We also included small stem density (SI4) as a variable because the prairie warbler is associated with dense understory vegetation. We used point count and habitat data reported by Annand and Thompson (1997) (Table 110) to derive a logistic function (Fig. 64) that predicted habitat suitability for the prairie warbler from small stem density.

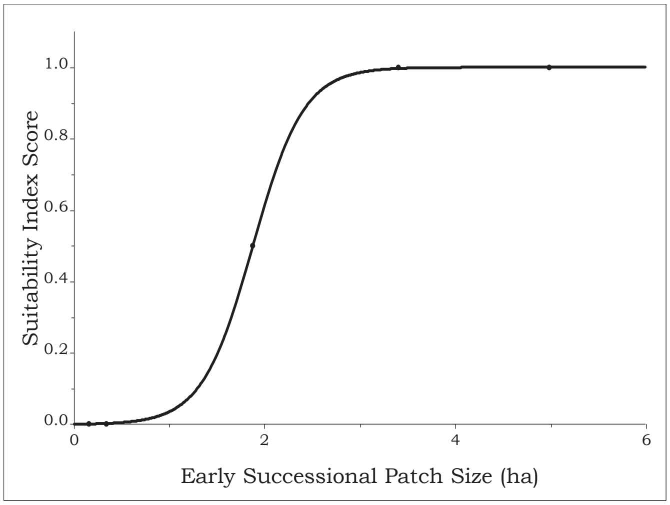


Figure 63.—Relationship between early successional patch size and suitability index (SI) scores for prairie warbler habitat. Equation: SI score = $(1.002 / (1 + (1207.332 * e^{-3.757 * \text{forest patch size}})))$.

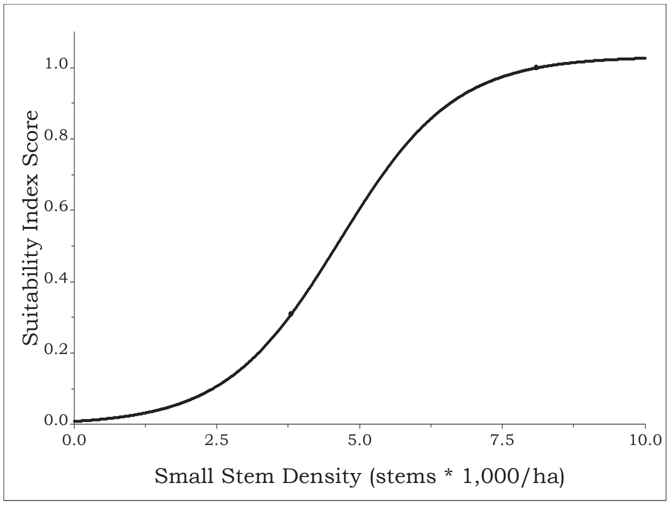


Figure 64.—Relationship between small stem (< 2.5 cm d.b.h.) density (stems * 1000/ha) and suitability index (SI) scores for prairie warbler habitat. Equation: SI score= $(1.000 / (1 + (99.749 * e^{-1.001 * (\text{small stem density} / 1000)})))$.

**Table 108.—Influence of edge on suitability index (SI) scores for prairie warbler habitat**

| 3 × 3 pixel window around early successional habitat includes mature forest[a] | SI score |
|---|---|
| Yes | 0.5 |
| No | 1.0 |

[a] Early successional = grass-forb, shrub-seedling, and sapling successional age classes; mature forest = pole or sawtimber successional age classes.

**Table 109.—Influence of early successional patch size on suitability index (SI) scores for prairie warbler habitat; early successional patches only include grass-forb, shrub-seedling, and sapling successional age classes**

| Early successional patch size (ha)[a] | SI score |
|---|---|
| 0.18 | 0.0 |
| 0.36 | 0.0 |
| 1.89 | 0.5 |
| 3.42 | 1.0 |
| 5.00 | 1.0 |

[a] Larson and others (2003).

**Table 110.—Influence of small stem (< 2.5 cm d.b.h.) density (stems * 1,000/ha) on suitability index (SI) scores for prairie warbler habitat**

| Small stem density | SI score |
|---|---|
| 0.0[a] | 0.00 |
| 3.8[b] | 0.31 |
| 8.1[b] | 1.00 |

[a] Assumed value.
[b] Annand and Thompson (1997).

Finally, we used data from Sheffield (1981) to inform an inverse logistic function (Fig. 65) that discounted SI scores at increasingly high canopy closures (SI5; Table 111).

To calculate the overall HSI score, we determined the geometric mean of SI scores for forest structure attributes (SI1, SI4, and SI5) and landscape composition (SI2 and SI3) separately and then the geometric mean of these means together.

Overall HSI = $((SI1 * SI4 * SI5)^{0.333} * (SI2 * SI3)^{0.500})^{0.500}$

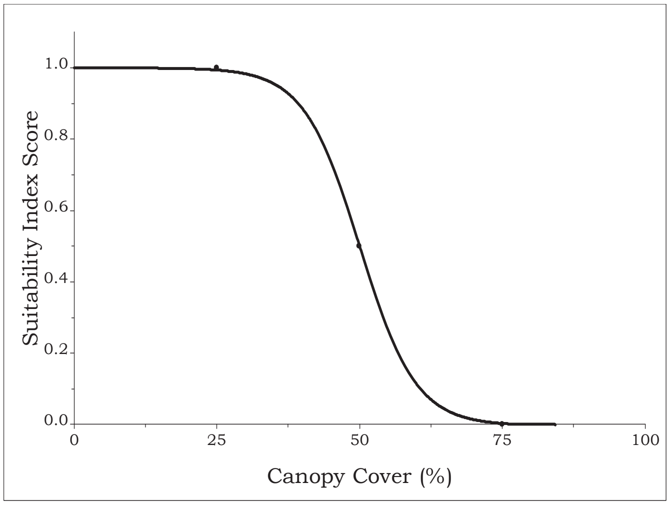


Figure 65.—Relationship between canopy cover and suitability index (SI) scores for prairie warbler habitat. Equation: SI score = 1 - (1.003 / (1 + (26950.420 * e $^{-0.204 * \text{canopy cover}}$))).

**Table 111.—Influence of canopy cover on suitability index (SI) scores for prairie warbler habitat**

| Canopy cover (percent)[a] | SI score |
|---|---|
| 0 | 1.0 |
| 25 | 1.0 |
| 50 | 0.5 |
| 75 | 0.0 |
| 100 | 0.0 |

[a] Sheffield (1981).

## Verification and Validation

The prairie warbler was found in all 88 subsections of the CH and WGCP. Spearman rank correlation identified a significant ($P \leq 0.001$) positive relationship ($r_s = 0.41$) between average HSI score and mean BBS route abundance across subsections. The generalized linear model predicting BBS abundance from BCR and HSI for the prairie warbler was significant ($P = 0.005$; $R^2 = 0.117$), and the coefficient on the HSI predictor variable was both positive ($\beta = 15.317$) and significantly different from zero ($P \leq 0.001$). Therefore, we considered the HSI model for the prairie warbler both verified and validated (Tirpak and others 2009a).

## Prothonotary Warbler

John and Karen Hollingsworth,
U.S. Fish & Wildlife Service

### Status

The prothonotary warbler (*Protonotaria citrea*) is a long-distance neotropical migrant associated with bottomland hardwood and floodplain forests of the Southeast. Densities are highest in the Mississippi Alluvial Valley; this species is notably absent from the central and southern Appalachians. Populations in the CH have remained relatively stable while those in the WGCP, where the prothonotary warbler is a Bird of Conservation Concern (Table 1), have declined by 5.8 percent per year since 1967 (Table 5). This bird is a planning and responsibility species in the CH (regional combined score = 14) and a management attention species in the WGCP (regional combined score = 17).

### Natural History

Because it nests in cavities and readily accepts nest boxes, the prothonotary warbler has been well-studied.

Petit (1999) provided an excellent, detailed description of this bird's habitat requirements:

> Key (and nearly universal) features are presence of water near wooded area with suitable cavity nest sites. Nest usually placed over or near large bodies of standing or slow-moving water, including seasonally flooded bottomland hardwood forest, baldcypress swamps, and large rivers or lakes (Walkinshaw 1953, Blem and Blem 1991). Many other forms of water also chosen, such as creeks, streams, backyard ponds, and even swimming pools. Nests located away from permanent water are usually in low-lying, temporarily flooded spots (Walkinshaw 1953).
>
> Other important habitat correlates include low elevation, flat terrain, shaded forest habitats with sparse understory, and in some places, presence of baldcypress (Kahl and others 1985, Robbins and others 1989). Common overstory trees in nesting habitat include willows, maples, sweet gum, willow oak, ashes, elms, river birch, black gum, tupelo, cypress, and other species associated with wetlands. Buttonbush is the most common subcanopy species. Canopy height 12-40 m (usually 16-20), canopy cover usually 50-75 percent; ground vegetation usually very sparse and of low stature (< 0.5 m; Kahl and others 1985).
>
> Exhibits area sensitivity, avoiding forests <100 ha in area and avoiding waterways with wooded borders <30 m wide (Kahl and others 1985).

### Model Description

The HSI model for prothonotary warbler includes seven variables: landform, landcover, successional age class, water, forest patch size, percentage of forest in the local (1-km radius) landscape, and snag density.

**Table 112.—Relationship of landform, landcover type, and successional age class to suitability index scores for prothonotary warbler habitat**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.100 | 0.300 | 0.400 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.100 | 0.300 | 0.400 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.300 | 0.800 | 1.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.200 | 0.600 | 0.800 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.200 | 0.600 | 0.800 |

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 112). We directly assigned SI scores to these combinations on the basis of relative rankings of habitat associations reported by Hamel (1992) for the prothonotary warbler in the Southeast.

This species is rarely found more than 200 m from water during the breeding season, so we used a 9 × 9 pixel window (270 x 270 m) to examine whether water was close enough to each site to make it suitable (SI2). If water was present in any of the 81 pixels comprising the window, we assigned the center pixel a value of 1.000. If water was absent, we assigned the center pixel a value of zero (Table 113).

We also included forest patch size (SI3) as a variable in the HSI model because prothonotary warbler abundance is lower in small isolated fragments and thin riparian buffer strips (Table 114; Fig. 66). However, this species occupies small forest fragments within heavily forested landscapes so we included the percentage of forest in the local landscape as a variable (SI4). To capture this relationship, we fit a logistic function (Fig. 67) to data (Table 115) derived

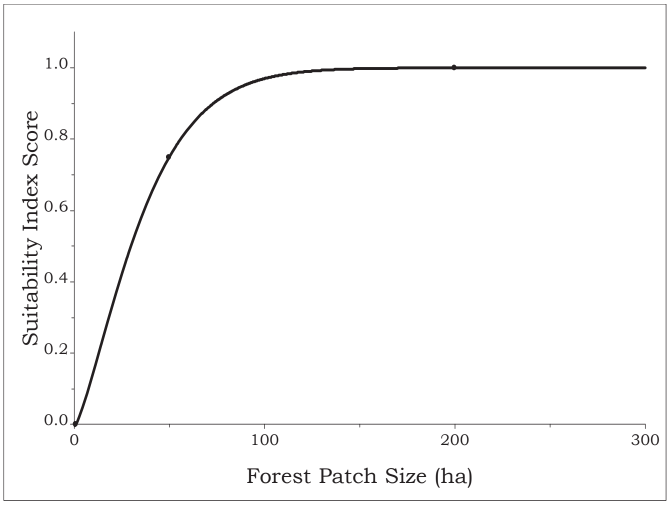


Figure 66.—Relationship between forest patch size and suitability index (SI) scores for prothonotary warbler habitat. Equation: $\text{SI score} = 1.002 - 1.001 * e^{-0.031 * (\text{forest patch size} \wedge 0.968)}$.

**Table 113.—Influence of occurrence of water on suitability index (SI) scores for prothonotary warbler habitat**

| 9 × 9 pixel window contains water | SI score |
|---|---|
| Yes | 1.0 |
| No | 0.0 |

**Table 114.—Influence of forest patch size on suitability index (SI) scores for prothonotary warbler habitat**

| Forest patch area (ha)[a] | SI score |
|---|---|
| 0 | 0.00 |
| 50 | 0.75 |
| 200 | 1.00 |
| 500 | 1.00 |

[a] Assumed value.

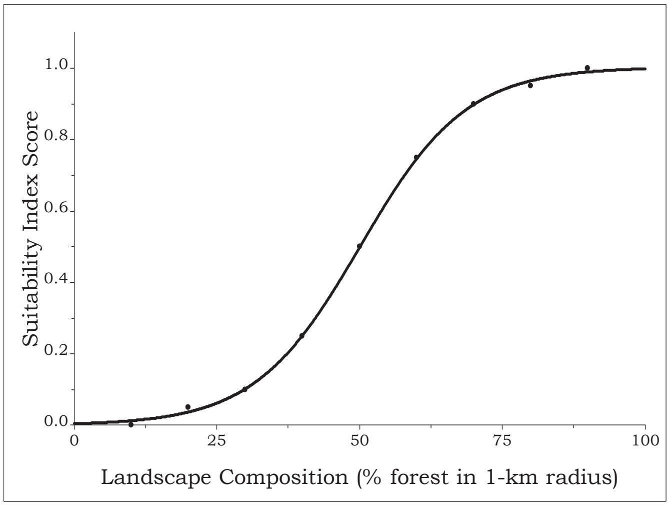


Figure 67.—Relationship between landscape composition and suitability index (SI) scores for prothonotary warbler habitat. Equation: $\text{SI score} = 1.005 / (1.000 + (221.816 * e^{-0.108 * (\text{landscape composition})}))$.

**Table 115.—Relationship between local landscape composition (percent forest in 1-km radius) and suitability index (SI) scores for prothonotary warbler habitat**

| Landscape composition | SI score |
|---|---|
| 0[a] | 0.00 |
| 10[a] | 0.00 |
| 20[a] | 0.05 |
| 30[b] | 0.10 |
| 40[a] | 0.25 |
| 50[b] | 0.50 |
| 60[a] | 0.75 |
| 70[b] | 0.90 |
| 80[a] | 0.95 |
| 90[b] | 1.00 |
| 100[a] | 1.00 |

[a] Assumed value.

[b] Donovan and others (1997).

from Donovan and others (1997), who observed differences in predator and brood parasite communities among highly fragmented (< 15 percent), moderately fragmented (45 to 50 percent), and lightly fragmented (> 90 percent forest) landscapes. We assumed that the midpoints between these classes (30 and 70 percent forest) defined the specific cutoffs for poor (SI score ≤ 0.10) and excellent (SI score ≥ 0.90) habitat, respectively. We applied the maximum value of SI3 or SI4 to all sites to compensate for the higher suitability of small forest blocks in predominantly forested landscapes.

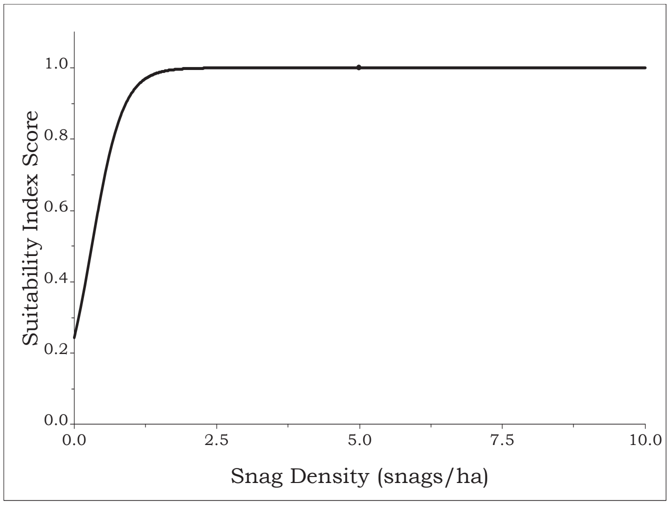


Figure 68.—Relationship between snag density and suitability index (SI) scores for prothonotary warbler habitat. Equation: SI score = $1.000 / (1 + (3.113 * e^{-3.689 * \text{snag density}}))$.

**Table 116.—Influence of snag density on suitability index (SI) scores for prothonotary warbler habitat**

| Snag density (snags/ha) | SI score |
|---|---|
| 0[a] | 0.25 |
| 5[b] | 1.00 |
| 20[a] | 1.00 |

[a] Assumed value.
[b] McComb and others (1986).

The prothonotary warbler is a cavity nester and uses snags (SI5) for nesting. McComb and others (1986) recommended 212 snags per 40 ha to satisfy the requirements of the primary cavity-nesting bird guild. We assumed that five snags per ha (Table 116) was sufficient for this bird (a secondary cavity-nesting species), but we recognized that this species also uses both cavities in live trees and crevices as nest sites. Therefore, we assigned a residual SI score (0.25) to sites lacking snags. We fit a logistic function through these points to quantify the snag density-habitat suitability relationship (Fig. 68).

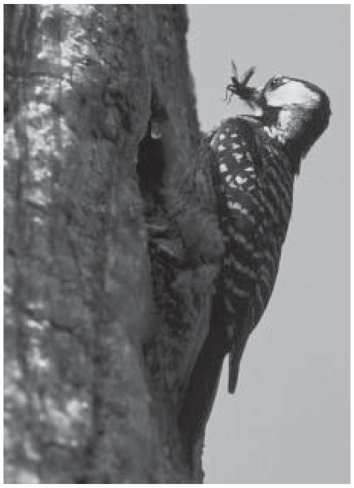
To calculate the overall HSI, we calculated the geometric mean of the two SIs related to forest structure (SI1 and SI5) and the product of the maximum of the two SIs related to landscape composition (SI3 or SI4) and SI2 separately and then the geometric mean of these values together.

Overall HSI = $((\text{SI1} * \text{SI5})^{0.500} * (\text{Max}(\text{SI3 or SI4}) * \text{SI2}))^{0.500}$

## Verification and Validation

The prothonotary warbler was found in 83 of the 88 subsections within the CH and WGCP. Spearman rank correlations identified significant positive associations between average HSI score and mean BBS route abundance across all subsections ($P \leq 0.001$; $r_s$ = 0.39) and subsections within which the prothonotary warbler were detected ($P \leq 0.001$; $r_s$ = 0.41). The generalized linear model predicting BBS abundance from BCR and HSI for the prothonotary warbler was significant ($P \leq 0.001$; $R^2 = 0.249$), and the coefficient on the HSI predictor variable was both positive ($\beta = 2.271$) and significantly different from zero ($P$ = 0.002). Therefore, we considered the HSI model for the prothonotary warbler both verified and validated (Tirpak and others 2009a).

## Red-cockaded Woodpecker


John and Karen Hollingsworth,
U.S. Fish & Wildlife Service

### Status

The red-cockaded woodpecker (*Picoides borealis*) is a federally endangered, nonmigratory resident of old-growth pine forest (particularly longleaf pine) throughout the Southeast (Jackson 1994). Due to the low detection rate for this species (0.05 bird/route in the WGCP), BBS data poorly estimates population trends (Table 5). The red-cockaded woodpecker is designated as a species warranting critical recovery in both the WGCP and CH (regional combined score = 21), though it is extirpated from the latter region.

### Natural History

Due to the limited availability of suitable habitat, the red-cockaded woodpecker lives in loose family groups and engages in cooperative breeding (Jackson 1994). Home ranges are large (average = 76.1 ha) but highly variable (17.2 to 159.5 ha; reviewed in Doster and James 1998).

Suitable habitat is defined by two primary habitat components. The first is the presence of large pines. Pines at least 35 cm d.b.h. generally are required for a stand to be occupied by the red-cockaded woodpecker (Davenport and others 2000, James and others 2001, Walters and others 2002). However, once large pine density exceeds 80 per ha, family group size (a demographic parameter related to productivity; Heppell and others 1994) declines (Walters and others 2002). Similarly, as the average d.b.h. of overstory pines increases above 35 cm, habitat quality declines (Davenport and others 2000), though these declines likely are linked to the maturation of the forests rather than to the negative effects of large trees directly. Similar patterns have been observed for overstory pine basal area and small pine tree density in occupied stands, where values for these habitat attributes are lower than local maxima (James and others 2001, Rudolph and others 2002, Walters and others 2002).

Open midstory is the second notable feature of high-quality habitat for the red-cockaded woodpecker. Hardwood midstory trees should be less than 3.26 m tall and ideally less than 1.8 m (Davenport and others 2002, Walters and others 2002). The open midstory typically is maintained through periodic fire (burn interval of 1 to 3 years), which also facilitates a wiregrass understory (James and others 2001). Because this species is nonmigratory and suitable habitat is disjunct, connectivity of patches is critical for the long-term persistence of this species across the landscape.

### Model Description

The HSI model for the red-cockaded woodpecker includes eight variables: landform, landcover, successional age class, forest patch size, pine basal area, hardwood basal area, connectivity, and large pine (> 35 cm d.b.h.) density.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 117).

**Table 117.—Relationship between landform, landcover type, age class, and suitability scores for red-cockaded woodpecker habitat; values in parentheses apply to West Gulf Coastal Plain/Ouachitas**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Evergreen | 0.000 | 0.000 | 0.200 | 0.600 | 0.800 |
| | Mixed | 0.000 | 0.000 | 0.200 | 0.400 | 0.400 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Evergreen | 0.000 | 0.000 | 0.200 | 0.600 | 0.800 |
| | Mixed | 0.000 | 0.000 | 0.200 | 0.400 | 0.400 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Evergreen | 0.000 | 0.000 | 0.200 | 0.600<br>(0.700) | 0.800<br>(1.000) |
| | Mixed | 0.000 | 0.000 | 0.200 | 0.400 | 0.400 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |

We directly assigned SI scores to these combinations on the basis of relative rankings of vegetation types and successional age classes for red-cockaded woodpeckers reported by Hamel (1992).

We included forest patch size (SI2) as a variable because of the large home ranges of the red-cockaded woodpecker. We assumed that the minimum and maximum home range sizes reported by Doster and James (1998) represented patch size thresholds for nonsuitable and optimal habitat, respectively. To inform the shape of the curve between these points, we assumed that the minimum area requirement of habitat identified in the red-cockaded woodpecker recovery plan (USDI Fish and Wildl. Serv. 2003) defined average (SI score = 0.500) habitat suitability. We used these data (Table 118) to define a logarithmic function to predict SI scores from forest patch size (Fig. 69).

Pine basal area (SI3) is a key component of red-cockaded woodpecker habitat, and sites with pine basal areas that are too low or too high are of poor quality. We fit a quadratic function (Fig. 70) to data from Conner and others (1995) and Walters and others (2002; Table 119) on the relative abundance of this species in habitats with varying levels of pine basal area.

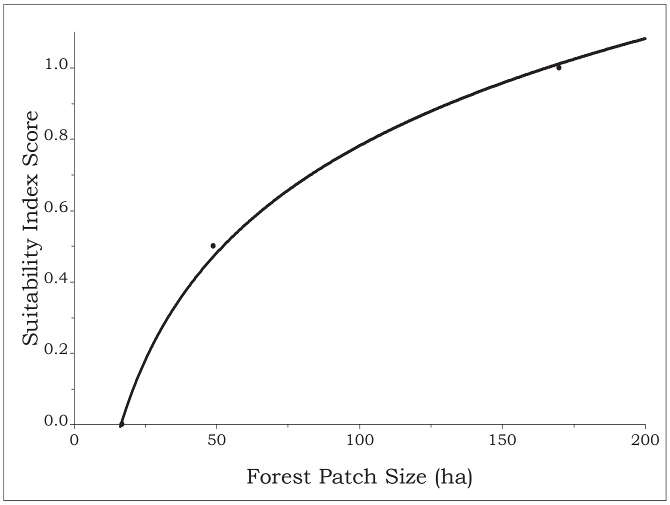


Figure 69.—Relationship between forest patch size and suitability index (SI) scores for red-cockaded woodpecker habitat. Equation: SI score = 0.4334 * ln(forest patch size) – 1.2133.

**Table 118.—Relationship between forest patch size and suitability index (SI) scores for red-cockaded woodpecker habitat**

| Forest patch size (ha) | SI score |
|---|---|
| 17[a] | 0.0 |
| 49[b] | 0.5 |
| 170[a] | 1.0 |

[a] Doster and James (1998).

[b] USDI Fish and Wildl. Serv. (2003).

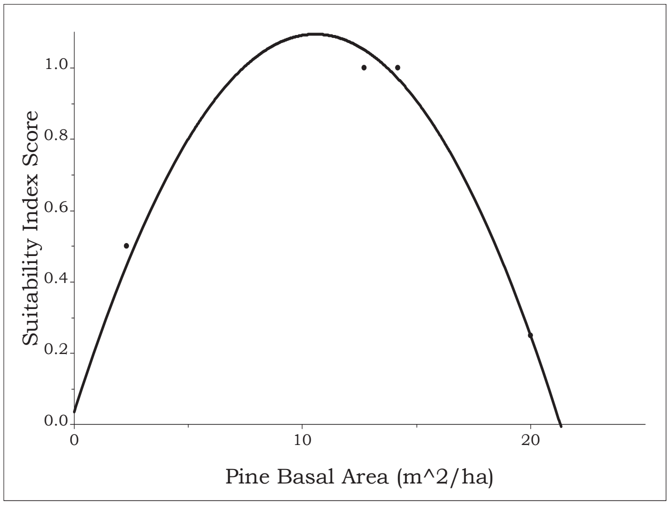


Figure 70.—Relationship between pine basal area and suitability index (SI) scores for red-cockaded woodpecker habitat. Equation: SI score = 0.0367 + 0.2006 * (pine basal area) – 0.009507 * (pine basal area)$^2$.

**Table 119.—Relationship between basal area of pines and suitability index (SI) scores for red-cockaded woodpecker habitat**

| Pine basal area ($m^2$/ha) | SI score |
|---|---|
| 0.0[a] | 0.00 |
| 2.3[b] | 0.50 |
| 12.7[c] | 1.00 |
| 14.2[c] | 1.00 |
| 20.0[a] | 0.25 |

[a] Assumed value.

[b] Walters and others (2000).

[c] Conner and others (1995).

Mid- and overstory hardwoods reduce habitat suitability for red-cockaded woodpeckers. We fit an inverse logistic function (Fig. 71) to data from Kelly and others (1993) and Wilson and others (1995) (Table 120) on the amount of hardwood basal area (SI4) around woodpecker nest cavities to predict habitat suitability based on this habitat feature.

As a resident species occupying disjunct habitat patches, the red-cockaded woodpecker exists in metapopulations. Therefore, dispersal between suitable forest patches is critical for the persistence of this species on the landscape. Isolated patches lacking a breeding female have no productivity, so we used the median dispersal distance for females (3.2 km; Jackson

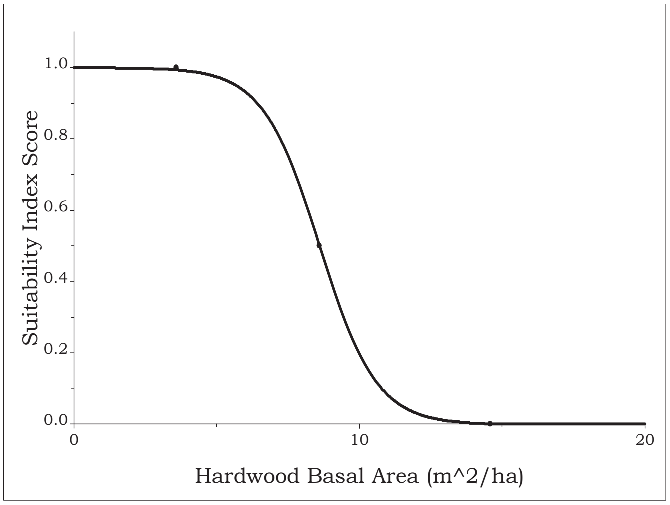


Figure 71.—Relationship between hardwood basal area and suitability index (SI) scores for red-cockaded woodpecker habitat. Equation: SI score = 1 - (1.001 / (1 + (5745.304 * e $^{-1.006\ *\ \text{hardwood basal area}}$))).

**Table 120.—Relationship between basal area of hardwoods ($m^2$/ha) and suitability index (SI) scores for red-cockaded woodpecker habitat**

| Hardwood basal area ($m^2$/ha) | SI score |
|---|---|
| 0.0[a] | 1.0 |
| 3.9[b] | 1.0 |
| 8.6[c] | 0.5 |
| 14.6[c] | 0.0 |
| 20.0[a] | 0.0 |

[a] Assumed value.

[b] Wison and others (1995).

[c] Kelly and others (1993).

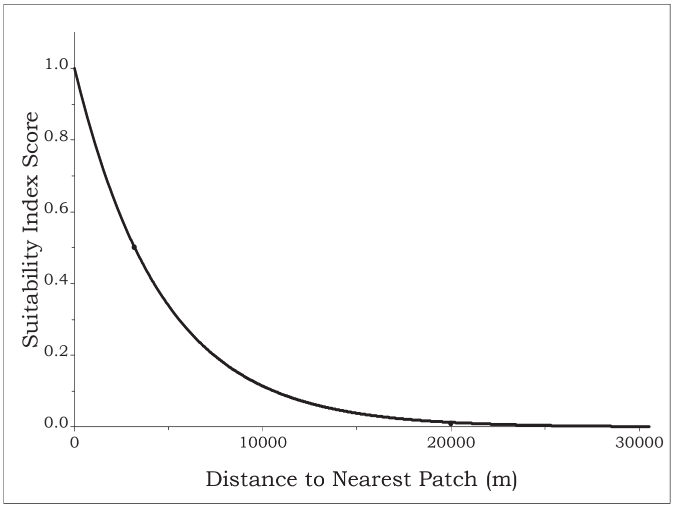


Figure 72.—Relationship between habitat connectivity and suitability index (SI) scores for red-cockaded woodpecker habitat. Equations: SI score = e $^{-0.0002\ *\ \text{distance to nearest habitat patch}}$.

**Table 121.—Relationship between distance to nearest habitat patch and suitability index (SI) scores for red-cockaded woodpecker habitat**

| Distance to nearest habitat patch (m) | SI score |
|---|---|
| 0[a] | 1.00 |
| 3,200[b] | 0.50 |
| 20,000[a] | 0.01 |

[a] Assumed value.

[b] Jackson (1994).

1994) to define average SI score (0.500). However, long-distance dispersal does occur (Larry Hedrick, 2006, U.S. Forest Service, pers. commun.), so we assigned to patches isolated more than 20 km from any other suitable site at least some residual suitability (0.010). We fit an exponential relationship (Fig. 72) through these data points (Table 121) to describe how the connectivity of patches influences habitat suitability.

Large pines (SI6) are a necessary component of red-cockaded woodpecker habitat because this bird disproportionately forages and nests in large pines. However, there is a threshold above which habitat suitability declines and increasingly large trees reduce the preferred open

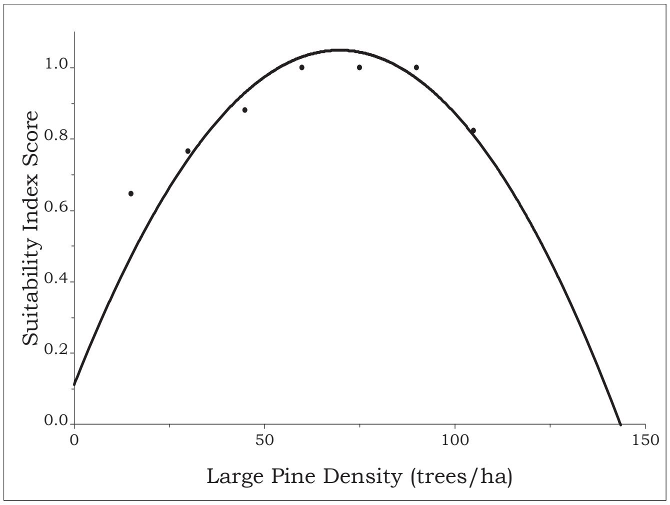


Figure 73.—Relationship between large pine tree (> 35 cm d.b.h.) density and suitability index (SI) scores for red-cockaded woodpecker habitat. Equation: SI score = 0.0269 * (pine tree density) – 0.000193 * (pine tree density)$^2$ + 0.1127.

**Table 122.—Relationship between large pine (> 35 cm d.b.h.) density (trees/ha) and suitability index (SI) scores for red-cockaded woodpecker habitat**

| Large pine density | SI score |
|---|---|
| 0[a] | 0.000 |
| 15[b] | 0.647 |
| 30[b] | 0.765 |
| 45[b] | 0.882 |
| 60[b] | 1.000 |
| 75[b] | 1.000 |
| 90[b] | 1.000 |
| 105[b] | 0.824 |

[a] Assumed value.
[b] Walters and others (2002).

character of the forest. We fit a quadratic function (Fig. 73) to data from Walters and others (2002), who identified this threshold at 60 to 90 large pines per ha (Table 122).

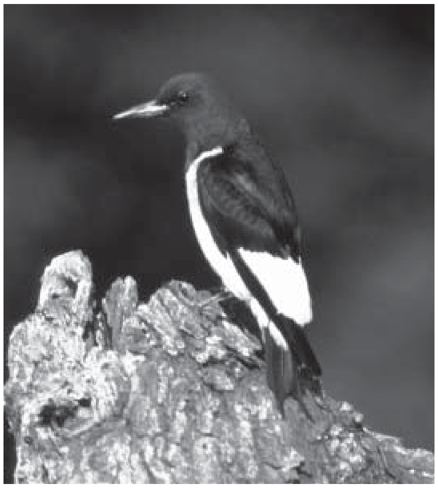
To calculate the overall HSI score, we determined the geometric mean of SI scores for forest structure (SI1, SI3, SI4, and SI6) and landscape composition (SI2 and SI5) separately and then the geometric mean of these means together.

Overall HSI = $((SI1 * SI3 * SI4 * SI6)^{0.250} * (SI2 * SI5)^{0.500})^{0.500}$

## Verification and Validation

The red-cockaded woodpecker was found in only 10 of the 88 subsections within the CH and WGCP. Spearman rank correlation identified a significant ($P \leq 0.001$) positive relationship ($r_s = 0.49$) between average HSI score and mean BBS route abundance across all subsections. However, when subsections where the red-cockaded woodpecker was not found were removed from the analysis, the relationship was not significant ($P = 0.645$; $r_s = 0.17$). Thus, the HSI model predicts the absence of the red-cockaded woodpecker better than its abundance in subsections where it is found. The generalized linear model predicting BBS abundance from BCR and HSI for the red-cockaded woodpecker was significant ($P \leq 0.001$; $R^2 = 0.203$), and the coefficient on the HSI predictor variable was both positive ($\beta = 0.094$) and significantly different from zero ($P = 0.042$). Therefore, we considered the HSI model for the red-cockaded woodpecker both verified and validated (Tirpak and others 2009a).

## Red-headed Woodpecker



Dave Menke, U.S. Fish & Wildlife Service

### Status

The red-headed woodpecker (*Melanerpes erythrocephalus*) is found throughout North America east of the Rocky Mountains; however, it is absent from New England and the higher elevations of the central and southern Appalachians. Since 1967, populations have declined by 3.2 percent per year in the WGCP and by 1 percent in the CH (Sauer and others 2005) (Table 5). This species is a Bird of Conservation Concern and a management attention priority in both the CH and WGCP (regional combined score = 16 and 17, respectively; Table 1).

### Natural History

The red-headed woodpecker is one of the most recognizable birds of the eastern United States and southern Canada, but few in-depth studies of this species have been conducted (Smith and others 2000). Nesting habitat consists of deciduous woodlands, including upland and bottomland hardwoods, riparian strips, open woods, open wooded swamps, groves of dead and dying trees, orchards, shelterbelts, parks, open agricultural lands, savannas, forest edges, roadsides, and utility poles (Smith and others 2000). It prefers xeric sites with large, tall trees, high basal area, and a sparse understory.

The red-headed woodpecker exhibits seasonal shifts in habitat use. Population dynamics are linked to annual fluctuations in oak acorn crops, and migration occurs in northern and western populations when hard mast is limited (Rodewald 2003). More locally, winter territories are established around small food caches within forest interiors; breeding territories are larger (3.1 to 8.5 ha in Florida) and concentrated along edges (Smith and others 2000).

Occurrence of the red-headed woodpecker varies with mean patch dimension, edge density of agricultural land, and the area of urban landcover (Lukomski 2003). It is a primary cavity excavator and snag availability may drive habitat selection (Giese and Cuthbert 2003). This species often is associated with high snag densities (Conner and others 1994) in mature stands near openings (Conner and Adkisson 1977, Brawn and others 1984). Snag density and basal area of dead elm distinguish nest sites from random sites in Minnesota (Giese and Cuthbert 2003). Similarly, loblolly pine stands with both standing and down dead woody debris removed contain fewer birds (Lohr and others 2002). Snags retained as groups provide multiple snags for roosting and foraging. Hardwood snags are used predominantly for foraging, whereas pine snags are more commonly used for nesting (Smith and others 2000). Thinnings and prescribed fires that open the understory and create snags are beneficial.

### Model Description

The HSI model for the red-headed woodpecker includes seven variables: landform, landcover, successional age class, snag density, large snag (> 20 cm d.b.h.) density, sawtimber tree (> 28 cm d.b.h.) density, and the occurrence of edge.

**Table 123.—Relationship of landform, landcover type, and successional age class to suitability index scores for red-headed woodpecker habitat; values in parentheses apply to West Gulf Coastal Plain/Ouachitas**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.125 | 0.250 | 0.250 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.125 | 0.250 | 0.250 |
| | Deciduous | 0.000 | 0.000 | 0.125 | 0.250 | 0.250 |
| | Evergreen | 0.000 | 0.000 | 0.250 | 0.500 | 0.500 |
| | Mixed | 0.000 | 0.000 | 0.250 | 0.500 | 0.500 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.125 | 0.250 | 0.250 |
| | Woody wetlands | 0.000 | 0.000 | 0.250 | 0.625 | 0.750 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.125 | 0.375 | 0.500 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.250 | 0.500 | 0.500 |
| | Deciduous | 0.000 | 0.000 | 0.125 | 0.375 | 0.500 |
| | Evergreen | 0.000 | 0.000 | 0.250 | 0.500 | 0.500 |
| | Mixed | 0.000 | 0.000 | 0.250 | 0.500 | 0.500 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.125 | 0.375 | 0.500 |
| | Woody wetlands | 0.000 | 0.000 | 0.250 | 0.500 | 0.500 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.250 | 0.750 | 1.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.250 | 0.500<br>(0.750) | 0.500<br>(1.000) |
| | Deciduous | 0.000 | 0.000 | 0.250 | 0.750 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.250 | 0.500<br>(0.750) | 0.500<br>(1.000) |
| | Mixed | 0.000 | 0.000 | 0.250 | 0.500 | 0.500 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.250 | 0.750 | 1.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.250 | 0.750 | 1.000 |

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 123). We directly assigned SI scores to these combinations on the basis of data from Hamel (1992) on the relative value of various vegetation types and successional age classes as red-headed woodpecker habitat in the Southeast.

This species relies heavily on snags for nesting, foraging, and roosting. King and others (2007) observed 31.8 snags per ha in savanna habitat used by the red-headed woodpecker, though basal area was only 0.9 $m^2$ per ha in that study. Therefore, we adjusted snag densities to reflect the intermediate basal area values (12 to 15 $m^2$/ha; Heltzel and Leberg 2006) characteristic of stands used by the red-headed woodpecker in the WGCP and CH BCRs. We assumed that 500 snags per ha represented an upper threshold above which maximal suitability was achieved and that 200 snags per ha represented a threshold below which sites were unsuitable (Table 124). We fit a logistic function (Fig. 74) through these data to predict how habitat suitability varied with snag density (SI2). Because the snag density in SI2 includes all dead trees greater than 2.5 cm d.b.h., we also included large snag (> 20 cm d.b.h.) density (SI3) as a variable. This additional requirement ensured the presence of snags suitable for nesting

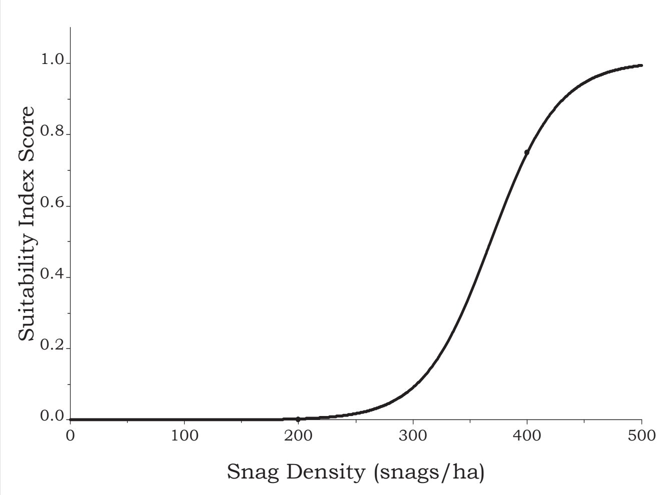


Figure 74.—Relationship between snag density (snags * 100/ha) and suitability index (SI) scores for red-headed woodpecker habitat. Equation: SI score = 1.006 / (1 + (249051.2 * e $^{(-0.0338 * \text{snag density})}$)).

**Table 124.—Influence of snag density on suitability index (SI) scores for red-headed woodpecker habitat**

| Snag density (snags/ha)[a] | SI score |
|---|---|
| 0 | 0.00 |
| 200 | 0.00 |
| 400 | 0.75 |
| 500 | 1.00 |

[a]Assumed value.

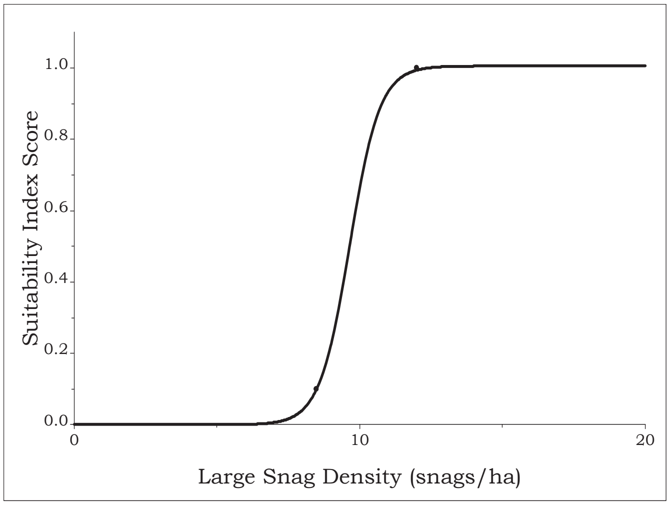


Figure 75.—Relationship between large snag (> 20 cm d.b.h.) density and suitability index (SI) scores for red-headed woodpecker habitat. Equation: SI score = 1.006 / (1 + (90614077 * e $^{(-1.899 * \text{large snag density})}$)).

**Table 125.—Influence of large snag (> 20 cm d.b.h.) density (snags/ha) on suitability index (SI) scores for red-headed woodpecker habitat**

| Large snag density | SI score |
|---|---|
| 0.0[a] | 0.0 |
| 8.5[b] | 0.1 |
| 12.0[a] | 1.0 |

[a]Assumed value.
[b]Lohr and others (2002).

in high-quality habitats. We relied on data from Lohr and others (2002) to inform an inverse logistic function (Fig. 75) that linked habitat suitability to large snag density (Table 125).

The red-headed woodpecker breeds in relatively open habitats with widely spaced large trees near openings (King and others 2007). Therefore, we included sawtimber tree density (SI4) and edge occurrence (SI5) as variables. We assumed that habitat suitability was highest when sawtimber tree density was 20 or fewer trees per ha and lowest when sawtimber tree density exceeded 50 trees per ha (Table 126). We fit a logistic function (Fig. 76) through these data points to quantify the relationship between sawtimber tree density and SI scores. To identify edges, we used a 7 × 7 pixel moving window (210 x 210 m) to locate the transitions between

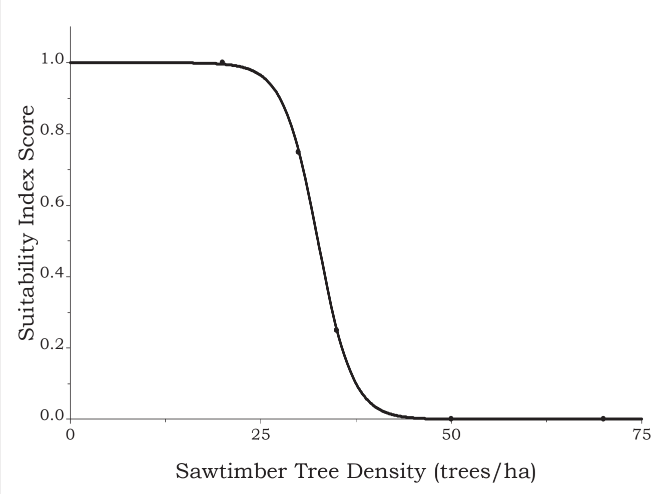


Figure 76.—Relationship between sawtimber tree (≥ 28 cm d.b.h.) density (trees * 10/ha) and suitability index (SI) scores for red-headed woodpecker habitat. Equation: SI score =1 – (1.000 / (1 + (1615169 * e $^{(-0.4398 * \text{sawtimber tree density})}$))).

**Table 126.—Influence of sawtimber tree (> 28 cm d.b.h.) density (trees/ha) on suitability index (SI) scores for red-headed woodpecker habitat**

| Sawtimber tree density[a] | SI score |
|---|---|
| 0 | 1.00 |
| 20 | 1.00 |
| 30 | 0.75 |
| 35 | 0.25 |
| 50 | 0.00 |
| 70 | 0.00 |

[a]Assumed value.

**Table 127.—Influence of edge on suitability index (SI) scores for red-headed woodpecker habitat**

| 7 × 7 window around forest pixel includes field[a] | SI score |
|---|---|
| Yes | 1.0 |
| No | 0.1 |

[a]Field defined as any shrub-seedling or grass-forb age class pixel, or natural grasslands, pasture-hay, fallow, urban-recreational grasses, emergent herbaceous wetlands, open water, high intensity residential, commercial-industrial-transportation, bare rock-sand-clay, quarries-strip mines-gravel pits, row crops, or small grains. Forest defined as any used sapling, pole, or sawtimber age class pixel of low-density residential, transitional, shrublands, deciduous, mixed, evergreen, orchard, or woody wetlands.

forest and non-forest landcovers or sapling-pole-sawtimber and grass-forb-shrub-seedling successional age class stands. We assigned to edge habitats the maximal SI score and discounted areas with no edge (Table 127).

To calculate the overall HSI score, we determined the geometric mean of SI scores for forest structure attributes (SI1, SI2, SI3, and SI4) and multiplied this product by the SI score for edge occurrence (SI5).

Overall HSI = ((SI1 * SI2 * SI3 * SI4)$^{0.250}$) * SI5

## Verification and Validation

The red-headed woodpecker was found in all 88 subsections of the CH and WGCP. Spearman rank correlation failed to identify a positive association between average HSI score and mean BBS abundance. The generalized linear model predicting BBS abundance from BCR and HSI for the red-headed woodpecker was significant ($P \leq 0.001$; $R^2 = 0.225$); however, the coefficient on the HSI predictor variable was negative ($\beta = -3.359$). Therefore, we considered the HSI model for the red-headed woodpecker neither verified nor validated (Tirpak and others 2009a).

## Swainson's Warbler

Chandler S. Robbins,
Patuxent Bird Identification InfoCenter
Photo used with permission

### Status

The Swainson's warbler (*Limnothlypis swainsonii*) is a neotropical migrant that breeds in dense thickets across the Southeast. Due to its overall low density and occurrence in habitats not well sampled by BBS, estimates of population trends based on this dataset are not reliable (Sauer and others 2005) (Table 5). Nonetheless, this species is a Bird of Conservation Concern and has a regional combined score of 20 in both the CH and WGCP (Table 1). An estimated 46 percent of the continental population of the Swainson's warbler breeds in the WGCP (Panjabi and others 2001).

### Natural History

The Swainson's warbler is distributed locally across the Southeast (Brown and Dickson 1994). Once believed to be restricted to canebrakes in bottomland hardwood and swamp forests of the Atlantic and Gulf Coastal Plains, it now has been documented breeding at low densities in regenerating clearcuts in Texas and rhododendron-mountain laurel thickets in the southern Appalachians (Graves 2002). Territory size is large for a wood warbler (3.2 ha) (Brown and Dickson 1994), and this species demonstrates area sensitivity. In Illinois, the Swainson's warbler is not observed on tracts smaller than 350 ha (Eddleman and others 1980).

This species does not use canopy height, basal area, successional age class, or species composition as habitat cues (Eddleman and others 1980, Graves 2002), but selects habitat based on understory characteristics. Dense thickets are required, and stem densities of about 35,000 stems per ha are optimal (Graves 2002). Canopy gaps are important for encouraging this dense growth, and canopy cover typically is high (70 to 80 percent) but rarely closed (> 90 percent) (Eddleman and others 1980, Graves 2001, Somershoe and others 2003). Understory vegetation is primarily woody; herbaceous cover is typically sparse (< 25 percent) (Eddleman and others 1980, Brown and Dickson 1994). Leaf litter is abundant and provides an important foraging substrate (Graves 2001, Somershoe and others 2003).

Hydrology is a critical factor influencing the habitat suitability for this warbler. In bottomland and floodplain habitats, birds select areas that typically are drier than surrounding sites (Graves 2001, Somershoe and others 2003). Inundation of otherwise suitable habitat from March - September negatively affects the quality of an otherwise suitable site (Graves 2002). This species occasionally breeds in xeric uplands with appropriate understory characteristics (Carrie 1996).

### Model Description

The HSI model for the Swainson's warbler includes six variables: landform, landcover, successional age class, forest patch size, proportion of forest in a 1-km radius, and small stem (< 2.5 cm d.b.h.) density.

**Table 128.—Relationship of landform, landcover type, and successional age class to suitability index scores for Swainson's warbler habitat**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.400 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.400 | 0.900 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.400 | 0.900 | 1.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.200 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.200 | 0.500 | 0.600 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.400 | 0.800 | 0.800 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.200 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.200 | 0.500 | 0.600 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.400 | 0.800 | 0.800 |

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 128). We adjusted the relative habitat quality rankings of Hamel (1992) for Swainson's warbler vegetation and successional age class associations to maximize habitat suitability in woody wetland habitats along floodplains, and to ensure that transitional sapling stands that may be used in the WGCP were assigned SI scores (Carrie 1996).

We included forest patch size (SI2) in the model because of the preference of the Swainson's warbler for interior sites within large forest tracts. We assumed that the minimum patch size in which Eddleman and others (1980) observed this species (350 ha) represented optimal habitat. Because this study was at the northern limit of the range of the Swainson's warbler, we assumed that birds would occupy significantly smaller tracts (Table 129). We based a logistic function on these assumptions to predict the impact of forest patch size on habitat suitability (Fig. 77). Nevertheless, the suitability of a specific patch size also is influenced by its landscape context (SI3). In predominantly forested landscapes, small forest patches that otherwise may not be suitable may be occupied due to their proximity to large forest blocks (Rosenberg and others 1999). To capture this relationship, we fit a logistic function (Fig. 78) to data (Table 130) derived from Donovan and others (1997), who observed differences

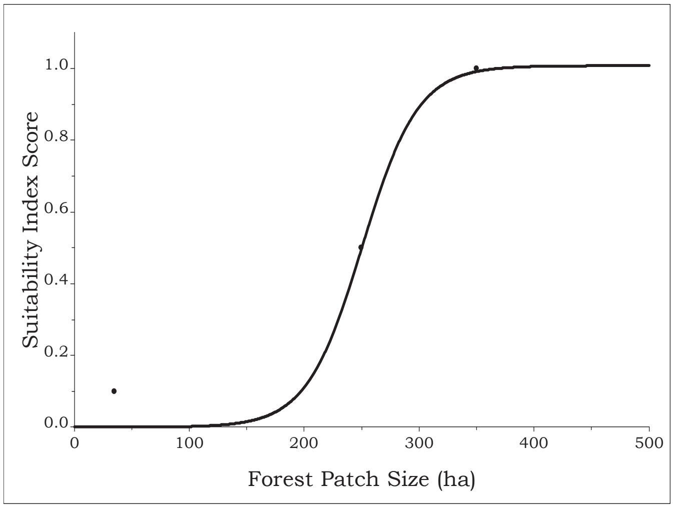


Figure 77.— Relationship between forest patch size and suitability index (SI) scores for Swainson's warbler habitat. Equation: SI score = $(1.001 / (1 + (31096.960 * e^{-0.041 * \text{(forest patch size)}})))$.

**Table 129.—Influence of forest patch size on suitability index (SI) score for Swainson's warbler habitat**

| Forest patch size (ha) | SI score |
|---|---|
| 0[a] | 0.00 |
| 35[a] | 0.01 |
| 250[a] | 0.50 |
| 350[b] | 1.00 |
| 500[a] | 1.00 |

[a]Assumed value.
[b]Eddleman and others (1980).

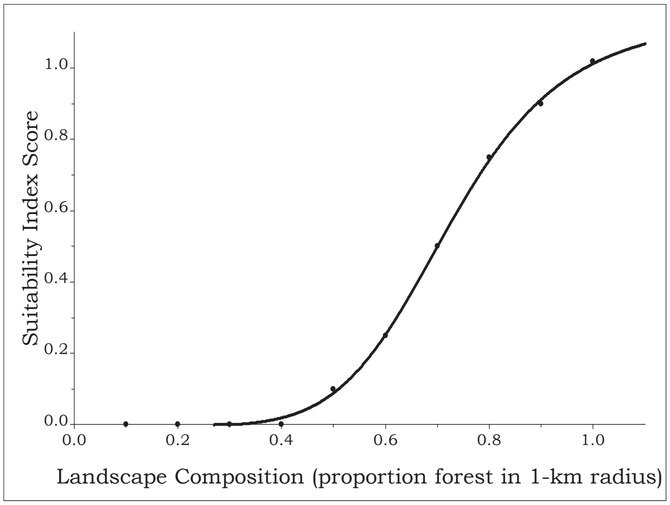


Figure 78.—Relationship between landscape composition and suitability index (SI) scores for Swainson's warbler habitat. Equation: SI score = $1.047 / (1.000 + (1991.516 * e^{-10.673 * \text{landscape composition}}))$.

**Table 130.—Relationship between landscape composition (proportion forest in 1-km radius) and suitability index (SI) scores for Swainson's warbler habitat**

| Landscape composition | SI score |
|---|---|
| 0.00[a] | 0.00 |
| 0.10[a] | 0.00 |
| 0.20[a] | 0.00 |
| 0.30[a] | 0.00 |
| 0.40[a] | 0.00 |
| 0.50[a] | 0.10 |
| 0.60[a] | 0.25 |
| 0.70[b] | 0.50 |
| 0.80[a] | 0.75 |
| 0.90[a] | 0.90 |
| 1.00[a] | 1.00 |

[a]Assumed value.
[b]Donovan and others (1997).

in predator and brood parasite communities among highly fragmented (< 15 percent), moderately fragmented (45 to 50 percent), and lightly fragmented (> 90 percent forest) landscapes. We assumed that the midpoint between moderately and lightly fragmented forest defined the specific cutoff for average (SI score = 0.500) habitat. We used the maximum score from SI2 or SI3 to account for the higher suitability of small patches in predominantly forested landscapes relative to their size alone.

The Swainson's warbler breeds in dense thickets and stem densities of approximately 35,000 stems per ha are optimal (SI score = 1.000) (Graves 2002). Stem densities can be even higher in early-successional bottomland hardwoods (> 200,000/ha), but we assumed habitat

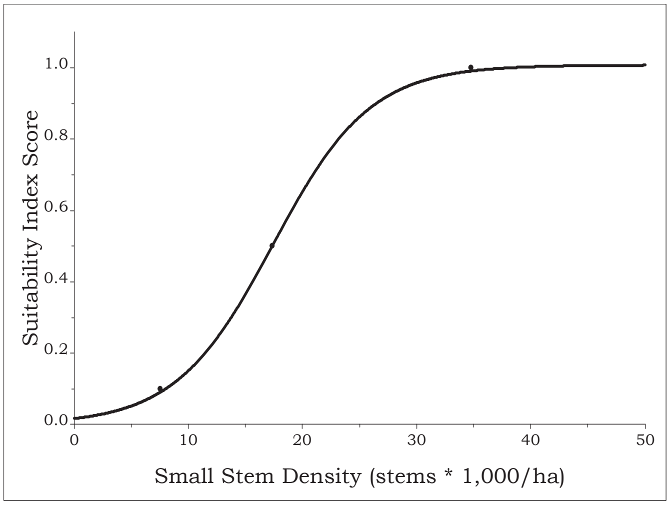


Figure 79.—Relationship between small stem (< 2.5 cm d.b.h.) density (stems * 1000/ha) and suitability index (SI) scores for Swainson's warbler habitat. Equation: SI score = 1.008 / (1.000 + (59.233 * $e^{-0.235 * (\text{small stem density} / 1000)}$)).

**Table 131.—Influence of small stem (< 2.5 cm d.b.h.) density (stems * 1,000/ha) on suitability index (SI) scores for Swainson's warbler habitat**

| Small stem density | SI score |
|---|---|
| 0.000[a] | 0.0 |
| 7.550[b] | 0.1 |
| 17.365[b] | 0.5 |
| 34.773[b] | 1.0 |
| 72.999[b] | 1.0 |

[a] Assumed value.
[b] Graves (2002).

suitability was not negatively affected by stem density. Therefore, we fit a logistic function (Fig. 79) to data from Graves (2002) that captured the effect of varying stem density on habitat suitability (Table 131).

To calculate the overall HSI score, we determined the geometric mean of SI scores for forest structure (SI1 and SI4) and multiplied that by the maximum SI score for forest patch size (SI2) or percent forest in the 1-km landscape (SI3) and finally calculated the geometric mean of that product.

Overall HSI = $((\text{SI1} * \text{SI4})^{0.500} * \text{Max}(\text{SI2 or SI3}))^{0.500}$

## Verification and Validation

The Swainson's warbler was found only in 31 of the 88 subsections within the CH and WGCP. Spearman rank correlation identified a significant ($P \leq 0.010$) positive relationship ($r_s = 0.31$) between average HSI score and mean BBS route abundance across all subsections. However, when subsections where this species was not found were removed from the analysis, the relationship was not significant ($P = 0.893$; $r_s = -0.03$). Thus, the HSI model better predicts the absence of the Swainson's warbler than its abundance in subsections where this species is found. The generalized linear model predicting BBS abundance from BCR and HSI for the Swainson's warbler was significant ($P \leq 0.001$; $R^2 = 0.260$); however, the coefficient on the HSI predictor variable was negative (ß = -0.298). Therefore, we considered the HSI model for the Swainson's warbler verified but not validated (Tirpak and others 2009a).

## Swallow-tailed Kite


D.A. Rintoul, Patuxent Bird Identification InfoCenter
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### Status

The swallow-tailed kite (*Elanoides forficatus*) is a neotropical raptor that reaches the northern limit of its distribution in the Unites States. Once ranging throughout the Mississippi River drainage as far north as Minnesota, this species now is restricted to seven states in the Southeast. There are too few swallow-tailed kites detected on BBS routes in the WGCP to estimate a population trend; however, this species is a Bird of Conservation Concern and immediate management attention priority in this BCR (regional combined score = 18; Table 1). The swallow-tailed kite no longer breeds in the CH and this species warrants critical recovery efforts in this region (regional combined score = 19).

### Natural History

The swallow-tailed kite is a rare breeder in the continental United States. The current restriction of this species to seven southern states (with limited distributions in all but Florida) represents a significant contraction of its former range. Most of the information on this bird in the United States is from Florida (Meyer 1995).

The swallow-tailed kite has a large home range (500 to 1800 ha) that increases substantially (> 20,000 ha) when the long but regular foraging forays characteristic of this species are included. With such a large home range, the important role of landscape structure on habitat suitability is not surprising. Critical habitat elements are large, tall trees for nesting and open habitats containing prey (Meyer 1995, Sykes and others 1999). Any interspersion of these features is useable (e.g., trees adjacent to prairie, wetlands, or marsh). Landscapes containing bottomland hardwood forest interspersed with scattered openings are particularly attractive. The edges of pine forests along swamps and riparian zones also are commonly used along the Coastal Plains. The Mississippi kite typically occupies habitats that are drier and contain more contiguous forest than the habitats of the swallow-tailed kite.

### Model Description

The HSI model for the swallow-tailed kite includes six variables: landform, landcover, successional age class, forest patch size, landscape composition, and dominant tree density.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 132). We then directly assigned SI scores to these combinations on the basis of relative habitat quality rankings from Hamel (1992) for the swallow-tailed kite. However, we assumed that only stands in the sawtimber successional age class provided suitable habitat for this species

We also included forest patch size (SI2) as a variable because of this bird's large home range and association with large blocks of forested wetlands. We fit a logarithmic function (Fig. 80)

**Table 132.—Relationship of landform, landcover type, and successional age class to SI scores for swallow-tailed kite habitat**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.000 | 0.500 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.500 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.500 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 1.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 0.800 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.000 | 0.800 |

to data (Table 133) from Zimmerman (2004) on the mean value of forest in 5-km buffers around swallow-tailed kite nest sites and the maximum home range size reported by Cely and Sorrow (1990) to assess the impact of forest patch size on habitat suitability scores for the swallow-tailed kite.

Like the Mississippi kite, the swallow-tailed kite forages aerially in open habitats, so it requires both forested sites for nesting and open areas for foraging (SI3). We based the ideal composition of vegetation types in the landscape on data from Sykes and others (1999), who observed 20 percent open habitat within 200-ha core areas in Florida. We maximized habitat suitability at this threshold and reduced SI scores in landscapes containing greater or lower proportions of open habitat (Table 134, Fig. 81).

The swallow-tailed kite nests in dominant trees (SI4) that extend above the canopy. We assumed that trees with a d.b.h. greater than 76.2 cm would extend above the canopy in the sawtimber stands that provide the exclusive habitat for this species. We assumed that one dominant tree per ha would satisfy this requirement and that the swallow-tailed kite would be absent from stands with a uniform canopy (zero dominant trees/ha). We fit an exponential function (Fig. 82) to the values between these data points and assumed that

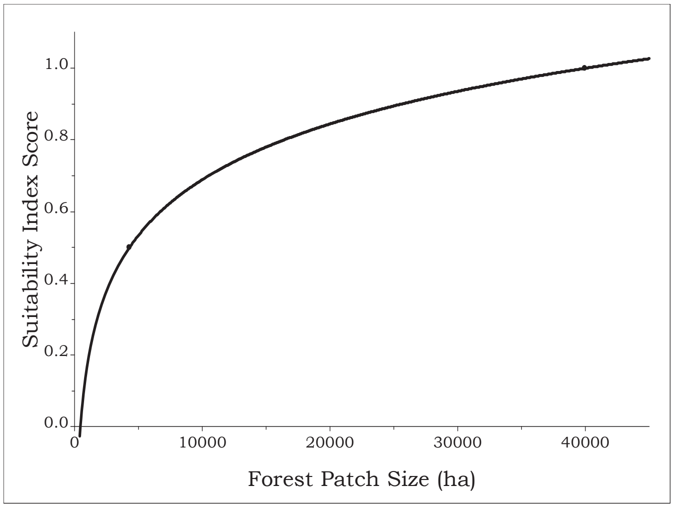

Figure 80.—Relationship between forest patch size and suitability index (SI) scores for swallow-tailed kite habitat. Equation: SI score = 0.224 * ln(forest patch size) – 1.376.

**Table 133.—Influence of forest patch size on suitability index (SI) scores for swallow-tailed kite habitat**

| Forest patch size (ha) | SI score |
|---|---|
| 4,300[a] | 0.5 |
| 40,000[b] | 1.0 |

[a] Zimmerman (2004).
[b] Cely and Sorrow (1990).

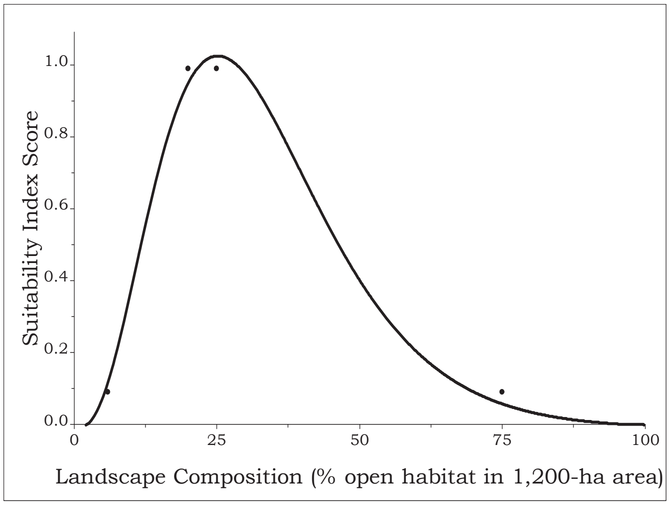

Figure 81.—Relationship between landscape composition and suitability index (SI) scores for swallow-tailed kite habitat. Equation: SI score = $(0.001 * 0.885^{(\text{percent open habitat})}) * (\text{percent open habitat})^{3.065}$.

**Table 134.—Suitability index scores for swallow-tailed kite habitat based on landscape composition (percent of open habitat) within 1,200-ha landscape**

| Landscape composition[a] | SI score |
|---|---|
| 6[b] | 0.1 |
| 20[c] | 1.0 |
| 25[b] | 1.0 |
| 75[b] | 0.1 |

[a] Water, grasslands, cultivated lands, and emergent wetlands.
[b] Assumed value.
[c] Sykes and others (1999).

stands with 14 dominant trees per ha (the maximum value from the WGCP during the FIA surveys of the 1990s) were associated with maximum habitat suitability (Table 135).

To calculate the overall HSI score, we determined the geometric mean of SI scores for forest structure attributes (SI1 and SI4) and landscape composition (SI2 and SI3) separately and then the geometric mean of these means together.

Overall HSI = $((SI1 * SI4)^{0.500} * (SI2 * SI3)^{0.500})^{0.500}$

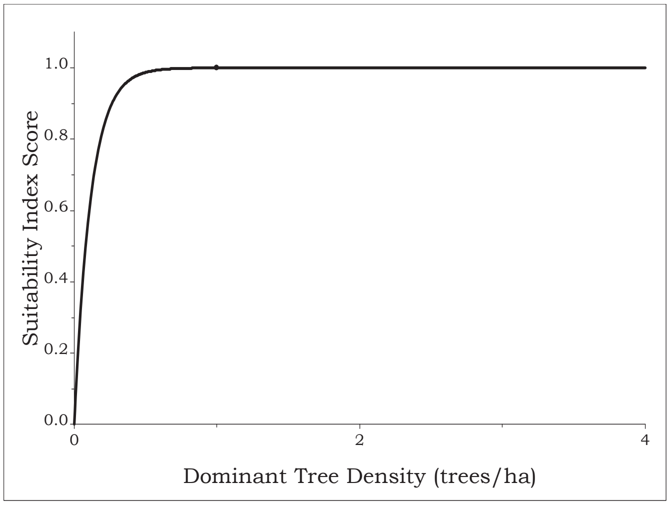

Figure 82.—Relationship between dominant tree density and (SI) scores for swallow-tailed kite habitat. Equation: SI score = $1 - e^{-8.734 * \text{dominant tree density}}$.

**Table 135.—Influence of dominant tree (> 76.2 cm d.b.h.) density (trees/ha) on suitability index (SI) scores for swallow-tailed kite habitat**

| Dominant tree density[a] | SI score |
|---|---|
| 0 | 0.0 |
| 1 | 1.0 |
| 14 | 1.0 |

[a] Assumed value.

## Verification and Validation

The swallow-tailed kite was found in 8 of the 88 subsections of the CH and WGCP. Spearman rank correlation identified a significant ($P \leq 0.001$) positive relationship ($r_s = 0.73$) between average HSI score and mean BBS route abundance across all subsections. However, when subsections where this species was not found were removed from the analysis, the relationship was not significant ($P = 0.432$; $r_s = 0.33$). Thus, the HSI model better predicts the absence of the swallow-tailed kite than its abundance in subsections where this species is found. The generalized linear model predicting BBS abundance from BCR and HSI for the swallow-tailed kite was significant ($P \leq 0.001$; $R^2 = 0.522$), and the coefficient on the HSI predictor variable was both positive ($\beta = 0.725$) and significantly different from zero ($P \leq 0.001$). Therefore, we considered the HSI model for the swallow-tailed kite both verified and validated (Tirpak and others 2009a).

## Whip-poor-will

Chandler S. Robbins,
Patuxent Bird Identification InfoCenter
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### Status

The whip-poor-will (*Caprimulgus vociferus*) is a neotropical migrant with a more northerly range than the chuck-will's-widow, though the ranges of the two are not exclusive and overlap broadly across the CH. The whip-poor-will has declined by 1.8 percent per year since 1967 in the CH (Sauer and others 2005) (Table 5), where this species is a Bird of Conservation Concern and has a regional combined score of 17 (Table 1). A large proportion of the continental population (35.5 percent) breeds in the CH (Panjabi and others 2001). This species is a rare breeder in the WGCP (regional combined score = 13).

### Natural History

Owing to its cryptic coloration and crepuscular activity pattern, the whip-poor-will is one of the least studied birds in North America (Cink 2002). Breeding habitat in the CH and WGCP consists of xeric deciduous and mixed forests with a sparse understory. This species also is associated with open areas, such as rural farmland, powerline and roadway rights-of-way, clearcuts and selectively logged forest, old fields, and reclaimed surface mines. Shaded forest stands with limited ground cover adjacent to open areas for foraging provide ideal whip-poor-will habitat. This species usually is absent from extensive areas of closed canopy forest, but there are no data on minimum or maximum thresholds for forest patch size. Small, isolated woodlots in a Maryland agricultural landscape are not used (Reese 1996, cited in Cink 2002). In Massachusetts, Grand and Cushman (2003) found that the whip-poor-will is strongly associated with complex patch shapes and high contrast edges. This species nests on the forest floor and hatching is synchronized with the full moon to optimize the foraging time of adults. Whip-poor-wills are not strongly territorial; home range varies from 2.8 to 11.1 ha.

### Model Description

The HSI model for whip-poor-will includes four variables: landform, landcover, successional age class, and the relative composition of forest and open habitats in the landscape.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 136). We directly assigned SI scores to these combinations on the basis of relative habitat rankings for vegetation and successional age class associations of the whip-poor-will reported by Hamel (1992).

The whip-poor-will nests in forest and forages in openings. As a result, it requires landscapes with an interspersion (SI2) of these landcover types. We assumed that a landscape with 70 percent forest and 30 percent open habitat was optimal (Michael Wilson, 2006, College of William & Mary, pers. commun.) and that landscapes with a greater proportion of forest

**Table 136.—Relationship of landform, landcover type, and successional age class to suitability index scores for whip-poor-will habitat**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.334 | 0.667 | 0.667 |
| | Deciduous | 0.000 | 0.000 | 0.334 | 0.667 | 0.667 |
| | Evergreen | 0.000 | 0.000 | 0.334 | 0.667 | 0.667 |
| | Mixed | 0.000 | 0.000 | 0.334 | 0.834 | 1.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.167 | 0.333 | 0.333 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.334 | 0.834 | 1.000 |
| | Deciduous | 0.000 | 0.000 | 0.334 | 0.667 | 0.667 |
| | Evergreen | 0.000 | 0.000 | 0.334 | 0.667 | 0.667 |
| | Mixed | 0.000 | 0.000 | 0.334 | 0.834 | 1.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.167 | 0.333 | 0.333 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.334 | 0.834 | 1.000 |
| | Deciduous | 0.000 | 0.000 | 0.334 | 0.667 | 0.667 |
| | Evergreen | 0.000 | 0.000 | 0.334 | 0.667 | 0.667 |
| | Mixed | 0.000 | 0.000 | 0.334 | 0.834 | 1.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.167 | 0.333 | 0.333 |

were more suitable than those with less forest cover so long as some openings were present (Table 137; sensu Cooper 1981).

We calculated the overall HSI score as the geometric mean of the two component variables.

Overall HSI = $(SI1 * SI2)^{0.500}$

## Verification and Validation

The whip-poor-will was found in 76 of the 88 subsections within the CH and WGCP. Spearman rank correlation identified a significant ($P = 0.005$) positive relationship ($r_s = 0.30$) between average HSI score and mean BBS route abundance across subsections. This relationship was even stronger ($r_s = 0.47$) when subsections in which the whip-poor-will was not detected were removed from the analysis. The generalized linear model predicting BBS abundance from BCR and HSI for the whip-poor-will was significant ($P = 0.002$; $R^2 = 0.139$), and the coefficient on the HSI predictor variable was positive ($\beta = 1.270$) but not significantly different from zero ($P = 0.229$). Therefore, we considered the HSI model for the whip-poor-will verified but not validated (Tirpak and others 2009a).

**Table 137.—Suitability index scores for whip-poor-will habitat based on the relative proportion of cells providing open and forest landcover within 500-m radius**

| Proportion forest[b] | Proportion open[a] | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | 0.0 | 0.1 | 0.2 | 0.3 | 0.4 | 0.5 | 0.6 | 0.7 | 0.8 | 0.9 | 1.0 |
| 0.0 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.0 |
| 0.1 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | |
| 0.2 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | | |
| 0.3 | 0.00 | 0.10 | 0.10 | 0.10 | 0.10 | 0.10 | 0.10 | 0.10 | | | |
| 0.4 | 0.00 | 0.25 | 0.25 | 0.25 | 0.25 | 0.25 | 0.25 | | | | |
| 0.5 | 0.00 | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | | | | | |
| 0.6 | 0.00 | 0.70 | 0.90 | 0.90 | 0.90 | | | | | | |
| 0.7 | 0.00 | 0.80 | 0.90 | 1.00 | | | | | | | |
| 0.8 | 0.00 | 0.80 | 0.90 | | | | | | | | |
| 0.9 | 0.00 | 0.80 | | | | | | | | | |
| 1.0 | 0.00 | | | | | | | | | | |

[a] Open = pasture/hay, recreational grasses, grasslands/herbaceous, and emergent herbaceous wetland landcovers or grass-forb and shrub-seedling successional age class stands.

[b] Forest = any habitats with positive SI1 values (Table 136).

## White-eyed Vireo

David Arbour, U.S. Forest Service

### Status

The white-eyed vireo (*Vireo griseus*) is a neotropical migrant that breeds throughout the southeastern United States. Populations have been stable in both the CH and WGCP over the last 40 years, but have been increasing in the WGCP by 1.6 percent annually since 1980 (Sauer and others 2005; Table 5). This species requires management attention in both the CH and WGCP (regional combined score = 15 and 16, respectively) but is not a Bird of Conservation Concern in either BCR (Table 1).

### Natural History

A small secretive songbird, the white-eyed vireo is associated with dense vegetation in secondary deciduous scrub-shrub, wood margins, overgrown pastures, abandoned farmlands, streamside thickets, and even mid- to late successional forests (Hopp and others 1995). This species shares habitats with the blue-gray gnatcatcher, Carolina wren, gray catbird, and brown thrasher, but prefers later successional forest than the yellow-breasted chat, prairie warbler, and Bell's vireo.

In Texas, the white-eyed vireo breeds in areas of shrubby vegetation (0 to 1 m) with dense foliage (Conner and Dickson 1997). Similarly, in Virginia, it prefers habitats with an extensive undergrowth of shrubs, brambles, and saplings interspersed with taller trees (10 to 20 percent of area). Vireo densities are higher in glade and regenerating forest habitat than edges in Missouri (Fink and others 2006). Densities also are inversely related to vegetation height, foliage density at 12 to 15 m, density of pole trees, and percent canopy closure (Conner and others 1983). Prather and Smith (2003) found that this species was more abundant in tornado-damaged forest in Arkansas than in undamaged areas. In South Carolina, abundance was positively related to gap size in bottomland forest that had been harvested by group-selection (Moorman and Guynn 2001). Territory size (0.1 to 1.8 ha) and population density vary with habitat quality. Brood parasitism affects nearly half of all nests and may significantly reduce productivity. The white-eyed vireo is more abundant in wide riparian strips of bottomland hardwood forest than in narrow strips (Kilgo and others 1998).

### Model Description

The HSI model for the white-eyed vireo includes six variables: landform, landcover, successional age class, edge occurrence, canopy cover, and small stem (< 2.5 cm d.b.h.) density.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 138). We directly assigned SI scores to these combinations on the basis of data from Hamel (1992) on the habitat associations of the white-eyed vireo in the Southeast.

**Table 138.—Relationship of landform, landcover type, and successional age class to suitability index (SI) scores for white-eyed vireo habitat**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 1.000 | 0.834 | 0.500 | 0.333 |
| | Deciduous | 0.000 | 1.000 | 0.834 | 0.500 | 0.333 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 1.000 | 0.834 | 0.500 | 0.333 |
| | Woody wetlands | 0.000 | 1.000 | 0.834 | 0.500 | 0.333 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.667 | 0.500 | 0.333 | 0.167 |
| | Deciduous | 0.000 | 0.667 | 0.500 | 0.333 | 0.167 |
| | Evergreen | 0.000 | 0.667 | 0.500 | 0.333 | 0.167 |
| | Mixed | 0.000 | 0.667 | 0.500 | 0.333 | 0.167 |
| | Orchard-vineyard | 0.000 | 0.667 | 0.500 | 0.333 | 0.167 |
| | Woody wetlands | 0.000 | 1.000 | 0.834 | 0.500 | 0.333 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.667 | 0.500 | 0.333 | 0.167 |
| | Deciduous | 0.000 | 0.667 | 0.500 | 0.333 | 0.167 |
| | Evergreen | 0.000 | 0.667 | 0.500 | 0.333 | 0.167 |
| | Mixed | 0.000 | 0.667 | 0.500 | 0.333 | 0.167 |
| | Orchard-vineyard | 0.000 | 0.667 | 0.500 | 0.333 | 0.167 |
| | Woody wetlands | 0.000 | 1.000 | 0.834 | 0.500 | 0.333 |

**Table 139.—Influence of edge on suitability index (SI) scores for white-eyed vireo habitat**

| 3 × 3 pixel window around forest pixel includes field? [a] | SI score |
|---|---|
| Yes[b] | 1.00 |
| No | 0.01 |

[a] Field defined as any sapling, shrub-seedling, or grass-forb age class pixel, or natural grasslands, pasture-hay, fallow, urban-recreational grasses, emergent herbaceous wetlands, open water, high-intensity residential, commercial-industrial-transportation, bare rock-sand-clay, quarries-strip mines-gravel pits, row crops, or small grains. Forest defined as any pole or sawtimber age class pixel of low-density residential, transitional, shrublands, deciduous, mixed, evergreen, orchard, or woody wetlands.

[b] Seedling-shrub and sapling habitats used regardless of edge.

In older forest stands, the white-eyed vireo concentrates on edges (SI2) and other areas with dense vegetation (Conner and Dickson 1997). We used a 3 × 3 pixel window (90 x 90 m) to identify the interfaces between pole and sawtimber successional age class forest and herbaceous and nonforest landcovers (hard edge) or shrub-seedling, grass-forb, and sapling successional age class forest (soft edge). We assumed that pole and sawtimber stands adjacent to these edges would have the highest SI score but applied a residual suitability value (0.01) to areas not identified as edge habitats to compensate for small forest gaps and openings that may be used. Shrub-seedling and sapling stands were suitable habitat regardless of edge (Table 139).

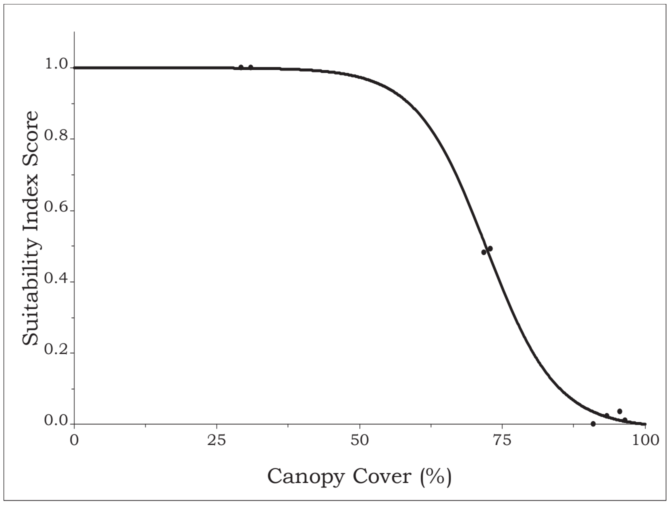

Figure 83.—Relationship between canopy cover and suitability index (SI) scores for white-eyed vireo habitat. Equation: SI score = 1 - (1.0101 / (1 + (127952.58 * $e^{-0.1629 * \text{canopy cover}}$))).

**Table 140.—Influence of canopy cover on suitability index (SI) scores for white-eyed vireo habitat**

| Canopy cover (percent) | SI score |
|---|---|
| 29.26[a] | 1.000 |
| 31.00[b] | 1.000 |
| 71.86[a] | 0.482 |
| 73.00[b] | 0.493 |
| 91.00[b] | 0.000 |
| 93.38[a] | 0.024 |
| 95.58[a] | 0.036 |
| 96.59[b] | 0.012 |

[a] Annand and Thompson (1997).

[b] Prather and Smith (2003).

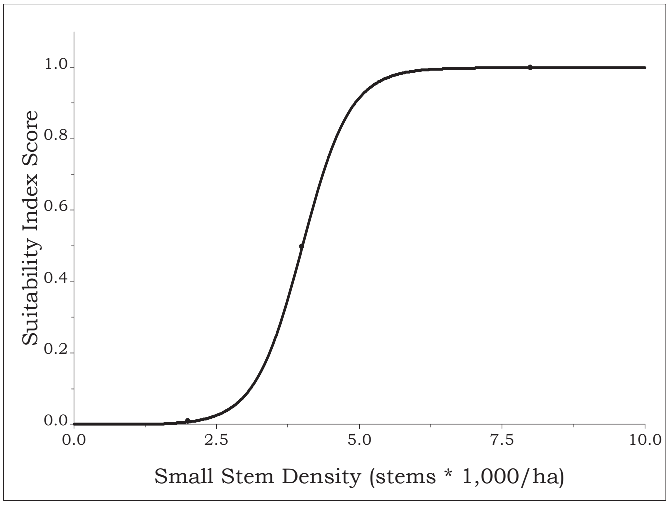

Figure 84.—Relationship between small stem (< 2.5 cm d.b.h.) density (stems * 1000/ha) and suitability index (SI) scores for white-eyed vireo habitat. Equation: SI score = (1.000 / (1 + (14512.121 * $e^{-2.396 * (\text{small stem density} / 1000)}$))).

**Table 141.—Influence of small stem (< 2.5 cm d.b.h.) density (stems * 1,000/ha) on suitability index (SI) scores for white-eyed vireo habitat**

| Small stem density[a] | SI score |
|---|---|
| 2 | 0.01 |
| 4 | 0.50 |
| 8 | 1.00 |

[a] Annand and Thompson (1997).

To refine the association of the white-eyed vireo with canopy gaps, we modeled the effect of canopy cover (SI3) on SI scores as an inverse logistic function (Fig. 83) that captured the absence of this species in closed-canopy forests (Table 140).

Finally, we fit a logistic function (Fig. 84) to data from Annand and Thompson (1997) (Table 141) on the influence of small stem (< 2.5 cm d.b.h.) density (SI4) on the relative density of the white-eyed vireo to quantify the relationship between SI scores and this habitat feature.

Assuming that this species uses edge as a surrogate to its preferred shrub-seedling and sapling habitats, we calculated HSI scores separately for shrub-seedling-sapling and pole-sawtimber forest stands. In the former, the geometric mean of forest structure variables alone defines the suitability score. For the latter, landscape composition (edge occurrence) also was a factor in the calculation.

Shrub-seedling and sapling (young) successional age classes:

$HSI_{Young}$: $(SI1 * SI3 * SI4)^{0.333}$

Pole and sawtimber (old) successional age classes:

$HSI_{Old}$: $((SI1 * SI3 * SI4)^{0.333} * SI2)^{0.500}$

To determine the overall HSI score, we summed the age class specific HSIs:

Overall HSI = $HSI_{Young} + HSI_{Old}$

## Verification and Validation

The white-eyed vireo was found in all 88 subsections of the CH and WGCP. Spearman rank correlation identified a significant ($P = 0.002$) positive association ($r_s = 0.33$) between average HSI score and mean BBS route abundance across all subsections. The generalized linear model predicting BBS abundance from BCR and HSI for the white-eyed vireo was significant ($P \leq 0.001$; $R^2 = 0.529$); however, the coefficient on the HSI predictor variable was negative ($\beta = -9.070$). Therefore, we considered the HSI model for the white-eyed vireo verified but not validated (Tirpak and others 2009a).

## Wood Thrush


Steve Maslowski, U.S. Fish & Wildlife Service

### Status

The wood thrush (*Hylocichla mustelina*) is a familiar woodland migrant to the forests of the eastern and central United States. Population declines for this species in the Midwest are linked to higher predation and parasitism rates in fragmented landscapes (Robinson and others 1995, Sauer and others 2005) (Table 5). The wood thrush is both a Bird of Conservation Concern and a management attention priority in the CH and WGCP (regional combined score = 16 and 15, respectively; Table 1).

### Natural History

The wood thrush is a long-distance neotropical migrant that exemplifies the decline in songbirds due to forest fragmentation. Due to its general abundance, ease of nest location and monitoring, and area sensitivity, the wood thrush is easy to study and there is a large body of knowledge on this bird (Roth and others 1996). This species is common in deciduous and mixed forests but rare in pure evergreen stands (Roth and others 1996). Mesic, upland forests with a moderate density of midcanopy trees and shrubs for nesting and an open understory with abundant leaf litter for foraging are optimal (Roth and others 1996). Closed overstory canopies are commonly used (Roth and others 1996, Bell and Whitmore 2000).

The wood thrush displays area sensitivity in productivity but not in its occupancy of habitats. It nests in forest fragments as small as 0.3 ha, albeit at low densities (Tilghman 1987, Weinberg and Roth 1998), and in narrow (< 150 m wide) riparian strips (Sargent and others 2003). However, nest predation and parasitism rates are extremely high in fragments of less than 80 ha and in riparian buffers less than 530 m wide (Donovan and others 1995, Hoover and others 1995, Peak and others 2004). Landscapes with greater amounts of forest cover (particularly unfragmented forest) mitigate some of these effects in small woodlots (Donovan and others 1997, Driscoll and Donovan 2004, Driscoll and others 2005). Nest success is predicted better by the amount of forest in the landscape than by the structural characteristics of microhabitat around nests (Hoover and Brittingham 1998, Driscoll and others 2005).

### Model Description

The HSI model for the wood thrush includes seven variables: landform, landcover, successional age class, forest patch size, percent forest in the local (1-km radius) landscape, small stem (< 2.5 cm d.b.h.) density, and canopy cover.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 142). We directly assigned SI scores to these combinations on the basis of habitat associations reported by Hamel (1992) but made minor adjustments to increase SI scores for sapling stands on the basis of data from Thompson and others (1992).

**Table 142.—Relationship of landform, landcover type, and successional age class to suitability index scores for wood thrush habitat; values in parentheses apply to West Gulf Coastal Plain/Ouachitas**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.250 | 0.750 | 0.750 | 1.000 |
| | Transitional-shrubland | 0.000 | 0.250 | 0.750 | 0.750 | 1.000 |
| | Deciduous | 0.000 | 0.250 | 0.750 | 0.750 | 1.000 |
| | Evergreen | 0.000 | 0.167 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.167 | 0.333 | 0.333 | 0.667 |
| | Orchard-vineyard | 0.000 | 0.250 | 0.333 | 0.333 | 0.667 |
| | Woody wetlands | 0.000 | 0.250 | 0.500 | 0.500 | 1.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.250 | 0.500 | 0.500 | 0.834 |
| | Transitional-shrubland | 0.000 | 0.167<br>(0.000) | 0.333<br>(0.000) | 0.333<br>(0.000) | 0.667<br>(0.000) |
| | Deciduous | 0.000 | 0.250 | 0.500 | 0.500 | 0.834 |
| | Evergreen | 0.000 | 0.167 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.167 | 0.333 | 0.333 | 0.667 |
| | Orchard-vineyard | 0.000 | 0.250 | 0.333 | 0.333 | 0.667 |
| | Woody wetlands | 0.000 | 0.334 | 0.667 | 0.667 | 1.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.334 | 0.667 | 0.667 | 1.000 |
| | Transitional-shrubland | 0.000 | 0.167<br>(0.000) | 0.333<br>(0.000) | 0.333<br>(0.000) | 0.667<br>(0.000) |
| | Deciduous | 0.000 | 0.334 | 0.667 | 0.500 | 0.667 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.167 | 0.333 | 0.333 | 0.667 |
| | Orchard-vineyard | 0.000 | 0.334 | 0.333 | 0.333 | 0.667 |
| | Woody wetlands | 0.000 | 0.334 | 0.667 | 0.667 | 1.000 |

Although the wood thrush will occupy small forest fragments, its density may be lower within them. Therefore, we included forest patch size (SI2) in the HSI model. We fit an exponential function (Fig. 85) to data from Robbins and others (1989) and Kilgo and others (1998) (riparian strips in this study were assumed to be 10 km long) that documented changes in relative occurrence with changes in patch size (Table 143). Nevertheless, the suitability of a forest patch is influenced not only by its size but also by its landscape context (SI3). To capture this relationship, we fit a logistic function (Fig. 86) to data (Table 144) derived from Donovan and others (1997), who observed differences in predator and brood parasite communities among highly fragmented (< 15 percent), moderately fragmented (45 to 50 percent), and lightly fragmented (> 90 percent forest) landscapes. We assumed that the midpoints between these classes (30 and 70 percent forest) defined the specific cutoffs for poor (SI score ≤ 0.10) and excellent (SI score ≥ 0.90) habitat, respectively. We used the maximum SI score from SI2 or SI3 to increase the suitability of small patches in heavily forested landscapes.

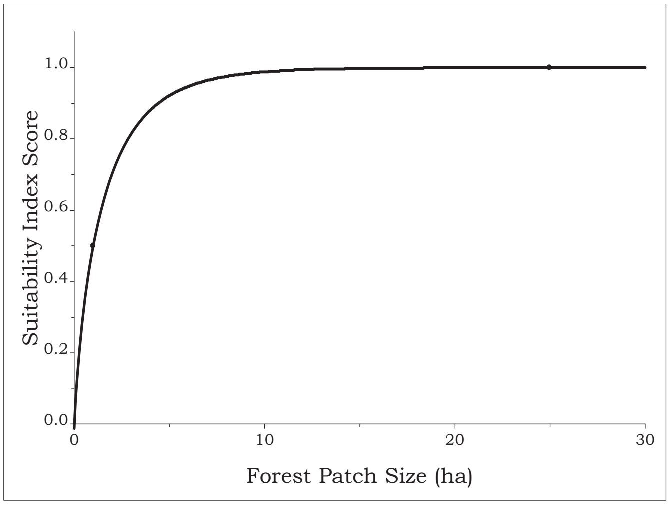

Figure 85.—Relationship between forest patch size and suitability index (SI) scores for wood thrush habitat. Equation: SI score = $1.000 – (1.017 * e^{-0.710 * (\text{forest patch size} \wedge 0.797)})$.

**Table 143.—Influence of forest patch size on suitability index (SI) scores for wood thrush habitat**

| Forest patch size (ha) | SI score |
|---|---|
| 0[a] | 0.0 |
| 1[a] | 0.5 |
| 25[b] | 1.0 |
| 500[a] | 1.0 |

[a] Robbins and others (1989).
[b] Kilgo and others (1998).

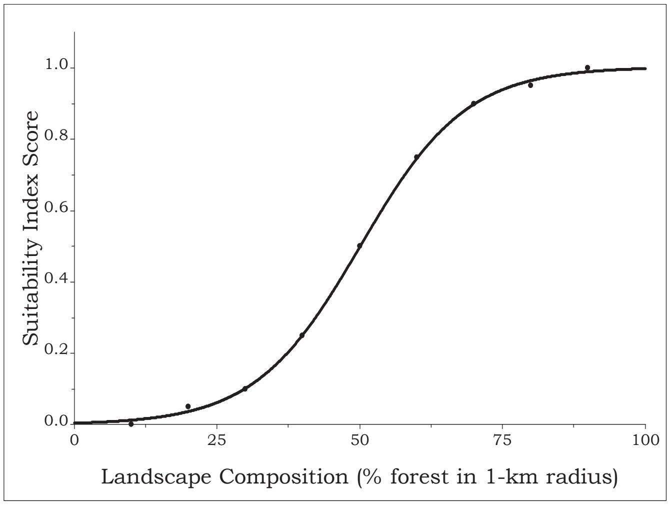

Figure 86.—Relationship between landscape composition and suitability index (SI) scores for wood thrush habitat. Equation: SI score = $1.005 / (1.000 + (221.816 * e^{-0.108 * \text{landscape composition}}))$.

**Table 144.—Relationship between landscape composition (percent forest in 1-km radius) and suitability index (SI) scores for wood thrush habitat**

| Landscape composition | SI score |
|---|---|
| 0[a] | 0.00 |
| 10[a] | 0.00 |
| 20[a] | 0.05 |
| 30[b] | 0.10 |
| 40[a] | 0.25 |
| 50[b] | 0.50 |
| 60[a] | 0.75 |
| 70[b] | 0.90 |
| 80[a] | 0.95 |
| 90[b] | 1.00 |
| 100[a] | 1.00 |

[a] Assumed value.
[b] Donovan and others (1997).

The wood thrush forages in leaf litter on the forest floor and is most common in stands with an open understory. We included small stem density (SI4) in the model as a proxy to understory cover. Although some researchers suggest that the wood thrush selects habitats with higher stem densities than generally are available, the controls in these studies typically are in mature forest and the wood thrush may simply be selecting habitats with locally high stem densities (Artman and Downhower 2003). We assumed that the average stem density (1,988 stems/ha) observed by Hoover and Brittingham (1998) around wood thrush nests was representative of optimal habitat. We discounted habitat suitability as small stem density increased due to presumed reductions in leaf litter, the preferred foraging substrate (Roth and others 1996). Nonetheless, Hoover and Brittingham (1998) observed wood thrush

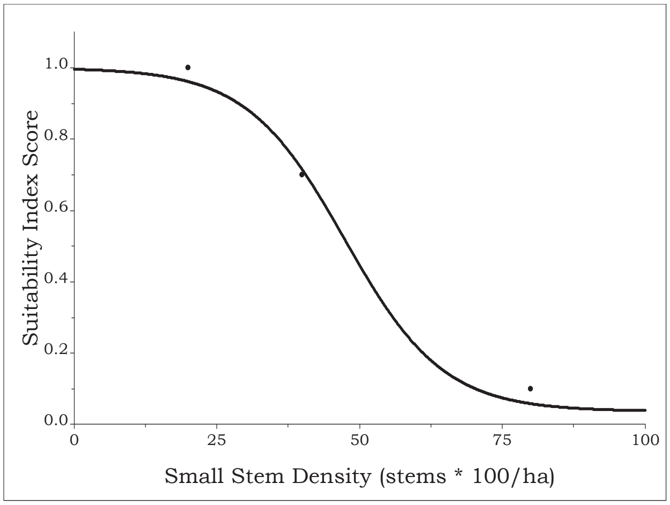

Figure 87.—Relationship between small stem (< 2.5 cm d.b.h.) density (stems * 100/ha) and suitability index (SI) scores for wood thrush habitat. Equation: SI score = 1 - (0.963 / (1 + (243.780 * $e^{-0.116 * (\text{small stem density} / 100)}$)).

**Table 145.—Influence of small stem (< 2.5 cm d.b.h.) density (stems * 100/ha) on suitability index (SI) scores for wood thrush habitat**

| Small stem density[a] | SI score |
|---|---|
| 0 | 1.0 |
| 20 | 1.0 |
| 40 | 0.7 |
| 80 | 0.1 |
| 100 | 0.0 |

[a]Assumed value.

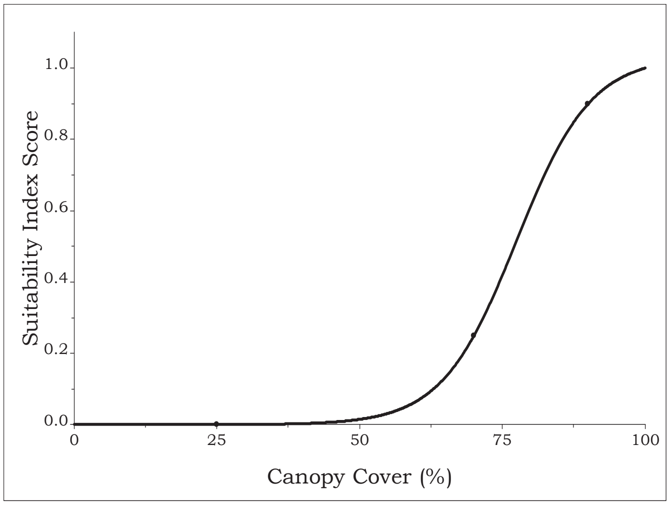

Figure 88.—Relationship between canopy cover and suitability index (SI) scores for wood thrush habitat. Equation: SI score = 1.032 / (1 + (141241.64 * $e^{-0.153 * \text{canopy cover}}$)).

**Table 146.—Influence of canopy cover (percent) on suitability index (SI) scores for wood thrush habitat**

| Canopy cover (percent) | SI score |
|---|---|
| 25[a] | 0.00 |
| 70[b] | 0.25 |
| 90[b] | 0.90 |
| 100[b] | 1.00 |

[a]Hoover and Brittingham (1998).

[b]Annand and Thompson (1997).

utilizing sites with extraordinarily high small stem densities (58,500 stems/ha, no doubt localized). Therefore, we assigned residual SI scores to sites with these characteristics. We fit an inverse logistic function (Fig. 87) to small stem density numbers that reflected this relationship (Table 145).

The wood thrush also is associated with closed-canopied forests, so we included canopy cover (SI5) as a variable and fit a logistic function (Fig. 88) to data from Annand and Thompson (1997) and Hoover and Brittingham (1998) to predict SI scores from canopy cover values (Table 146).

To calculate the overall HSI score, we determined the geometric mean of SI scores for forest structure attributes (SI1, SI4, and SI5) and then calculated the geometric mean of this value and the maximum of SI scores from forest patch size or percent forest in the landscape (Max(SI2 or SI3)).

$$\text{Overall HSI} = ((\text{SI1} * \text{SI4} * \text{SI5})^{0.333} * \text{Max(SI2 or SI3)})^{0.500}$$

## Verification and Validation

The wood thrush was found in all 88 subsections of the CH and WGCP. Spearman rank correlation identified a significant ($P \leq 0.001$) positive relationship ($r_s = 0.52$) between average HSI score and mean BBS route abundance across subsections. The generalized linear model predicting BBS abundance from BCR and HSI for the wood thrush was significant ($P \leq 0.001$; $R^2 = 0.311$), and the coefficient on the HSI predictor variable was both positive ($\beta = 9.992$) and significantly different from zero ($P \leq 0.001$). Therefore, we considered the HSI model for the wood thrush both verified and validated (Tirpak and others 2009a).

## Worm-eating Warbler

Charles H. Warren, images.nbii.gov

### Status

The worm-eating warbler (*Helmitheros vermivorus*) breeds on forested slopes of the eastern deciduous forest. It is notably absent from the Mississippi floodplain and the relatively flat forest-prairie ecotone immediately east of the Great Plains. Its preference for rugged terrain and its high-pitched, insect-like song result in underestimations of its density from roadside surveys. As a result, there are no credible trends from BBS data for this species (Table 5). Nevertheless, this species is a Bird of Conservation Concern in both BCRs. However, PIF designates the worm-eating warbler as a management attention priority in the CH (regional combined score = 18) and a planning and responsibility species in the WGCP (regional combined score = 15; Table 1).

### Natural History

The worm-eating warbler is a neotropical migrant that breeds in forest interiors of the Eastern United States (Hanners and Patton 1998). Minimum area requirements range from 21 ha in the mid-Atlantic (Robbins and others 1989) to more than 800 ha in Missouri (Wenny and others 1993). This species nests on the ground along moderate to steep slopes (≥ 20 percent) with dense (≥ 48 percent) shrub understories in mature deciduous and mixed deciduous-coniferous forests (Gale and others 1997). Both Artman and others (2001) and Blake (2005) found that the worm-eating warbler was less abundant in recently burned stands due to the loss of leaf litter, a preferred nesting and foraging substrate. Canopy closure exceeded 95 percent in both Missouri (Wenny and others 1993) and Connecticut (Gale and others 1997).

### Model Description

The HSI model for the worm-eating warbler includes seven variables: landform, landcover, successional age class, slope, forest patch size, percent forest in the landscape, and small stem (< 2.5 cm d.b.h.) density.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 147). We directly assigned SI scores to these combinations on the basis of habitat associations reported by Hamel (1992).

We included slope (SI2) in our model because of the prevalence of steep slopes in the territories of the worm-eating warbler. We defined slope classes on the basis of data from Gale and others (1997) who identified the relative preference of various slopes for this species (Table 148).

We also included forest patch size (SI3) as a variable to account for the preference of the worm-eating warbler for forest interiors. We fit a modified exponential function (Fig. 89) to data from Robbins and others (1989) to quantify the relationship between patch size

**Table 147.—Relationship of landform, landcover type, and successional age class to suitability index scores for worm-eating warbler habitat**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.300 | 0.700 | 0.800 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.200 | 0.400 | 0.400 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.200 | 0.500 | 0.600 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.300 | 0.800 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.200 | 0.400 | 0.400 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.200 | 0.400 | 0.400 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.200 | 0.600 | 0.800 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.200 | 0.400 | 0.400 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.200 | 0.400 | 0.400 |

and habitat suitability (Table 149). The suitability of a forest patch is influenced by its size and landscape context (SI4). To capture this relationship, we fit a logistic function (Fig. 90) to data (Table 150) derived from Donovan and others (1997), who observed differences in predator and brood parasite communities among highly fragmented (< 15 percent), moderately fragmented (45 to 50 percent), and lightly fragmented (> 90 percent forest) landscapes. We assumed that the midpoints between these classes (30 and 70 percent forest) defined the specific cutoffs for poor (SI score ≤ 0.10) and excellent (SI score ≥ 0.90) habitat, respectively. We assigned the maximum SI score of SI3 or SI4 to each site to account for the higher suitability of small forest patches in heavily forested landscapes.

We relied on data from Wenny and others (1993) and Annand and Thompson (1997) (Table 151) to quantify the relationship between SI scores and small stem density (SI5; Fig. 91). We assumed that the worm-eating warbler occupied forests with low stem densities, but these sites had lower suitability scores than sites with well developed understories characterized by dense stems.

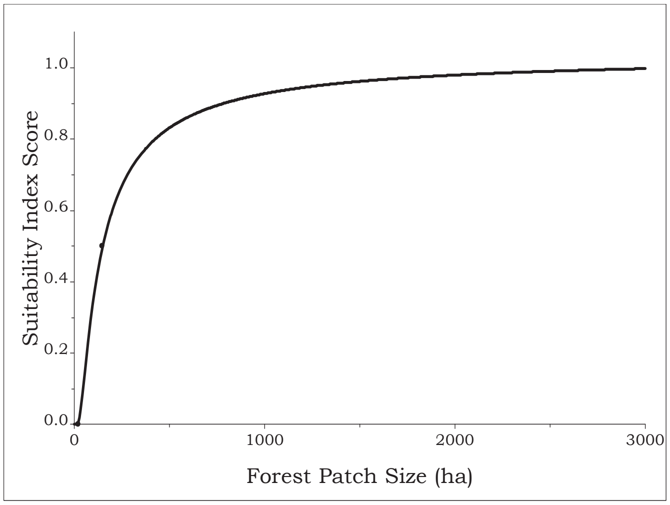

Figure 89.—Relationship between forest patch size and suitability index (SI) scores for worm-eating warbler habitat. Equation: SI score = $1.035 * e^{-109.238 / (\text{forest patch size})}$.

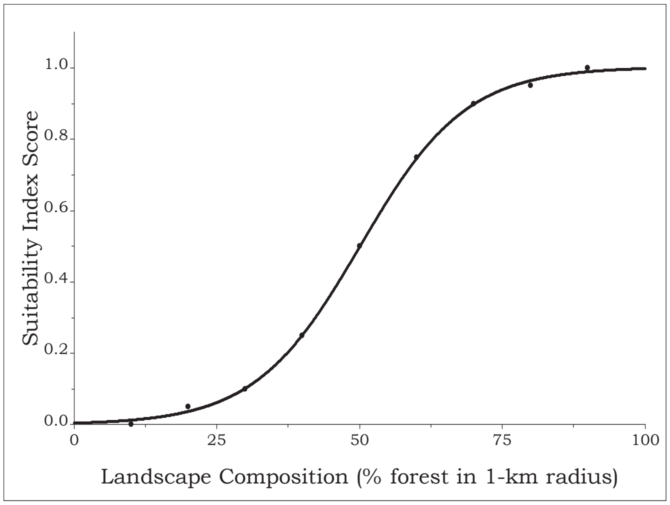

Figure 90.—Relationship between landscape composition and suitability index (SI) scores for worm-eating warbler habitat. Equation: SI score = $1.005 / (1.000 + (221.816 * e^{-0.108 * (\text{landscape composition})}))$.

**Table 148.—Influence of slope on suitability index (SI) scores for worm-eating warbler habitat**

| Slope (percent)[a] | SI score |
|---|---|
| < 5 | 0.0 |
| 5-20 | 0.5 |
| 21 | 1.0 |

[a] Gale and others (1997).

**Table 149.—Influence of forest patch size on suitability index (SI) scores for worm-eating warbler habitat**

| Forest patch size (ha) | SI score |
|---|---|
| 21[a] | 0.0 |
| 120[b] | 0.5 |
| 3,200[a] | 1.0 |

[a] Robbins and others (1989).

[b] Assumed value.

**Table 150.—Relationship between landscape composition (percent forest in 1-km radius) and suitability index (SI) scores for worm-eating warbler habitat**

| Landscape composition | SI score |
|---|---|
| 0[a] | 0.00 |
| 10[a] | 0.00 |
| 20[a] | 0.05 |
| 30[b] | 0.10 |
| 40[a] | 0.25 |
| 50[b] | 0.50 |
| 60[a] | 0.75 |
| 70[b] | 0.90 |
| 80[a] | 0.95 |
| 90[b] | 1.00 |
| 100[a] | 1.00 |

[a] Assumed value.

[b] Donovan and others (1997).

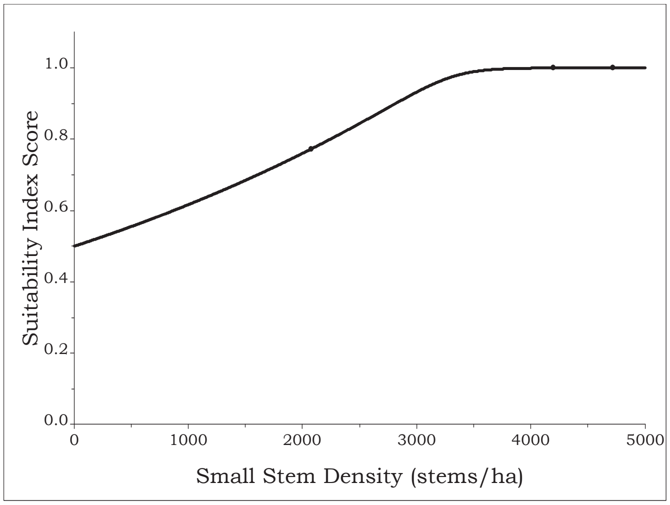

Figure 91.—Relationship between small stem (< 2.5 cm d.b.h.) density and suitability index (SI) scores for worm-eating warbler habitat.
Equation: SI score = $1.000 / (1 + e^{18.707 - 0.006 * (\text{small stem density})})^{\wedge 1 / 26.989}$
Equation takes the general form: $y = a/(1 + e^{b-cx})^{1/d}$.

**Table 151.—Influence of small stem (< 2.5 cm d.b.h.) density (stems/ha) on suitability index (SI) scores for worm-eating warbler habitat**

| Small stem density | SI score |
|---|---|
| 0[a] | 0.500 |
| 2,077[b] | 0.773 |
| 4,200[c] | 1.000 |
| 4,717[b] | 1.000 |

[a] Assumed value.
[b] Annand and Thompson (1997).
[c] Wenny and others (1993).

To calculate the overall HSI score, we determined the geometric mean of SI scores for forest structure (SI1 and SI5) and landscape composition (Max(SI3 or SI4) and SI2) separately and then the geometric mean of these means together.

Overall HSI = $((\text{SI1} * \text{SI5})^{0.500} * (\text{Max(SI3 or SI4)} * \text{SI2})^{0.500})^{0.500}$

## Verification and Validation

The worm-eating warbler was found in all 88 subsections of the CH and WGCP. Spearman rank correlation identified a significant ($P \leq 0.001$) positive relationship ($r_s = 0.66$) between average HSI score and mean BBS route abundance across subsections. The generalized linear model predicting BBS abundance from BCR and HSI for the worm-eating warbler was significant ($P \leq 0.001$; $R^2 = 0.408$), and the coefficient on the HSI predictor variable was both positive ($\beta = 1.798$) and significantly different from zero ($P \leq 0.001$). Therefore, we considered the HSI model for the worm-eating warbler both verified and validated (Tirpak and others 2009a).

## Yellow-billed Cuckoo

U.S. Forest Service

### Status

The yellow-billed cuckoo (*Coccyzus americanus*) is a neotropical migrant that breeds throughout North America east of the Rocky Mountains. The yellow-billed cuckoo is abundant in the CH and WGCP (10.43 and 12.93 birds/route, respectively), but populations in these BCRs have declinded slightly (Table 5). Although the yellow-billed cuckoo is not a Bird of Conservation Concern in either BCR, it is a management attention priority in both due to the importance of these regions (the core of this bird's range) for the sustainability of the continental population (Table 1).

### Natural History

A long-distance migrant, the yellow-billed cuckoo breeds in low, dense scrub near streams, marshes, and wetlands within otherwise open woodlands (Hughes 1999). It is among the most common birds in floodplain habitats along the Mississippi River and occupies both young cottonwood-willow stands and mature silver maple forests (Knutson and others 2005). This species exhibits some area sensitivity. Conner and others (2004) found that the yellow-billed cuckoo was most abundant in riparian strips more than 70 m wide, and Aquilani and Brewer (2004) recorded highest abundances in forest tracts larger than 55 ha.

Breeding success is correlated with insect outbreaks, particularly those of hairy caterpillars, and population densities vary greatly with food supply. Nests are located in dense, broad-leaved, deciduous shrubs or trees within 10 m of the ground. Twedt and others (2001) reported no difference in nest success between bottomland hardwoods and cottonwood plantations, nor did Wilson (1999) report a difference in nest success among stands subject to alternative thinning rates in Arkansas. On the basis of anticipated harvest scenarios, Klaus and others (2005) predicted that populations of the yellow-billed cuckoo would decline by approximately 37 percent on the Cherokee National Forest over the next 60 years.

### Model Description

The HSI model for the yellow-billed cuckoo includes seven variables: landform, landcover, successional age class, edge occurrence, midstory tree (11 to 25 cm d.b.h.) density, percent forest in the landscape (10-km radius), and forest patch size.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 152). We directly assigned SI scores to these combinations on the basis of habitat associations of the yellow-billed cuckoo reported by Hamel (1992). We increased SI scores within floodplain-valley and terrace-mesic landforms to account for the higher abundance of the yellow-billed cuckoo on these sites in the CH and WGCP.

Table 152.—Relationship of landform, landcover type, and successional age class to suitability index scores for yellow-billed cuckoo habitat. Values in parentheses apply to West Gulf Coastal Plain/Ouachitas

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.500 | 0.667 | 1.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.500 | 0.667 | 1.000 |
| | Deciduous | 0.000 | 0.000 | 0.500 | 0.667 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.167 | 0.333 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.500 | 0.667 | 1.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.333 | 0.667 | 0.667 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.500 | 0.667 | 1.000 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.500<br>(0.000) | 0.667<br>(0.000) | 1.000<br>(0.000) |
| | Deciduous | 0.000 | 0.000 | 0.500 | 0.667 | 1.000 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.167 | 0.333 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.500 | 0.667 | 1.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.333 | 0.667 | 0.667 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.250 | 0.333 | 0.500 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.250<br>(0.000) | 0.333<br>(0.000) | 0.500<br>(0.000) |
| | Deciduous | 0.000 | 0.000 | 0.250 | 0.333 | 0.500 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.083 | 0.167 | 0.167 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.250 | 0.333 | 0.500 |
| | Woody wetlands | 0.000 | 0.000 | 0.167 | 0.333 | 0.333 |

This species is more abundant within edge (SI2) habitats than within forest interiors (Kroodsma 1984). We used a 9 × 9 pixel moving window (270 x 270 m) to identify habitat edges and assumed that these locations represented optimal habitat. Nevertheless, nonedge habitats also are used by the yellow-billed cuckoo so we assigned to these sites only a slightly lower SI score (0.667; Table 153).

The yellow-billed cuckoo breeds in forest stands with well-developed midstories (SI3). We fit a quadratic function (Fig. 92) to data from Annand and Thompson (1997) on the relative densities of this species in stands with different midstory tree densities (Table 154) to predict how SI scores responded to changes in this habitat variable.

Table 153.—Influence of edge on suitability index (SI) scores for yellow-billed cuckoo habitat

| 9 × 9 pixel window around forest pixel includes field[a] | SI score |
|---|---|
| Yes | 1.000 |
| No | 0.667 |

[a] Field defined as any shrub-seedling or grass-forb age class pixel, or natural grasslands, pasture-hay, fallow, urban-recreational grasses, emergent herbaceous wetlands, open water, high-intensity residential, commercial-industrial-transportation, bare rock-sand-clay, quarries-strip mines-gravel pits, row crops, or small grains. Forest defined as any used sapling, pole, or sawtimber age class pixel of low-density residential, transitional, shrublands, deciduous, mixed, evergreen, orchard, or woody wetlands.

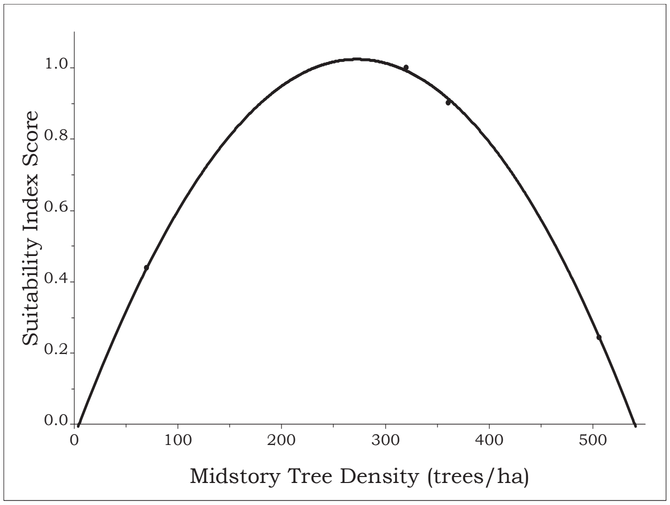

Figure 92.—Relationship between midstory tree (11–25 cm d.b.h.) density and suitability index (SI) scores for yellow-billed cuckoo habitat. Equation: SI score = 0.0078 * (midstory tree density) – 0.00001 * (midstory tree density)$^2$ – 0.0355.

**Table 154.—Influence of midstory tree (11–25 cm d.b.h.) density (trees/ha) on suitability index (SI) scores for yellow-billed cuckoo habitat**

| Midstory tree density[a] | SI score |
|---|---|
| 70 | 0.439 |
| 320 | 1.000 |
| 361 | 0.902 |
| 506 | 0.244 |

[a] Annand and Thompson (1997).

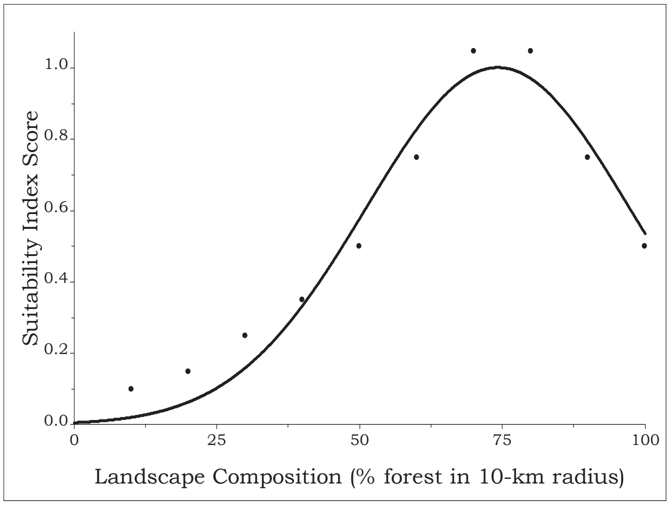

Figure 93.—Relationship between landscape composition and suitability index (SI) scores for yellow-billed cuckoo habitat. Equation: SI score = 1.002 * $e^{((0 – ((\text{landscape forest composition} * 100) – 74.165) \wedge 2) / 1064.634)}$.

**Table 155.—Relationship between landscape composition (percent forest in 10-km radius) and suitability index (SI) scores for yellow-billed cuckoo habitat**

| Landscape composition[a] | SI score |
|---|---|
| 0 | 0.00 |
| 10 | 0.10 |
| 20 | 0.20 |
| 30 | 0.30 |
| 40 | 0.40 |
| 50 | 0.50 |
| 60 | 0.75 |
| 70 | 1.00 |
| 80 | 1.00 |
| 90 | 0.75 |
| 100 | 0.50 |

[a] Assumed value.

Although a forest-breeding species, the yellow-billed cuckoo is associated with fragmented landscapes (Robbins and others 1989, Hughes 1999). We assumed that 70 to 80 percent forest in a 10-km landscape (SI4) was characteristic of ideal habitat (Table 155) and fit a function that reduced SI scores symmetrically as forest compositions departed from these ideal proportions (Fig. 93). Nevertheless, the cuckoo exhibits area sensitivity and may be absent or at low densities in small fragments (Robbins and others 1989, Bancroft and others 1995, Hughes 1999). Therefore, we used data from these sources to derive a logistic function (Fig. 94) that quantified the relationship between habitat suitability and forest patch size (SI5; Table 156).

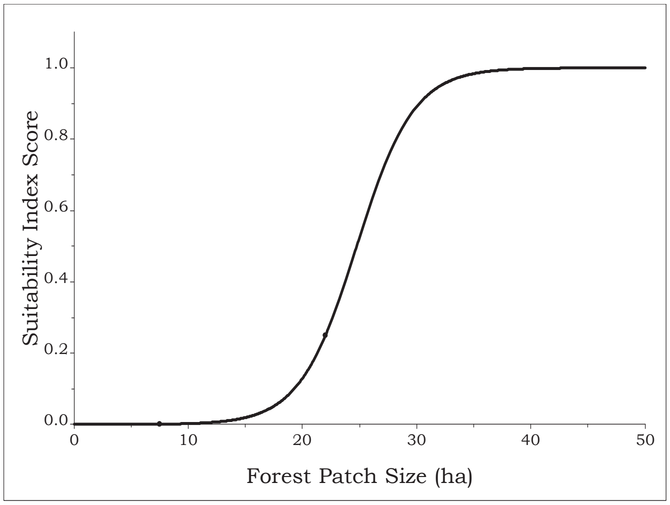

Figure 94.—Relationship between forest patch size and suitability index (SI) scores for yellow-billed cuckoo habitat. Equation: SI score = 1.000 / (1.000 + (20350.850 * e $^{-0.401 * \text{forest patch size}}$)).

**Table 156.—Influence of forest patch size on suitability index (SI) scores for yellow-billed cuckoo habitat**

| Forest patch size (ha) | SI score |
|---|---|
| 0[a] | 0.00 |
| 7.5[b] | 0.00 |
| 22[c] | 0.25 |
| 50[d] | 1.00 |

[a] Assumed value.

[b] Bancroft and others (1995).

[c] Hughes (1999).

[d] Robbins and others (1989).

To calculate the overall HSI score, we determined the geometric mean of SI scores for forest structure (SI1 and SI3) and landscape composition (SI2, SI4, and SI5) separately and then the geometric mean of these means together.

$$\text{Overall HSI} = ((\text{SI1} * \text{SI3})^{0.500} * (\text{SI2} * \text{SI4} * \text{SI5})^{0.333})^{0.500}$$

## Verification and Validation

The yellow-billed cuckoo was found in all 88 subsections of the CH and WGCP. Spearman rank correlation identified a significant ($P = 0.024$) positive relationship ($r_s = 0.24$) between average HSI score and mean BBS route abundance across subsections. The generalized linear model predicting BBS abundance from BCR and HSI for the yellow-billed cuckoo was significant ($P \leq 0.001$; $R^2 = 0.190$), and the coefficient on the HSI predictor variable was positive ($\beta = 5.265$) but not significantly different from zero ($P = 0.302$). Therefore, we considered the HSI model for the yellow-billed cuckoo verified but not validated (Tirpak and others 2009a).

## Yellow-breasted Chat

Chandler S. Robbins, Patuxent Bird Identification InfoCenter
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### Status

The yellow-breasted chat (*Icteria virens*) is a neotropical migrant that breeds in early successional habitats across the eastern United States. The distribution of this species in the West is patchy. Populations have responded to the loss of early successional habitat and have declined sharply across the northern edge of this bird's distribution (Sauer and others 2005). Within the CH, where this species has a regional combined score of 16 and is a management attention priority, populations have declined by approximately 2 percent per year during the last 40 years (Table 5). Conversely, at the southern limit of their range, populations have increased (1.3 percent annual increases in the WGCP from 1966 to 2005; Table 5).

### Natural History

The yellow-breasted chat breeds in low, dense, deciduous and evergreen vegetation within forests lacking a closed canopy (Eckerle and Thompson 2001). Habitat associations include forest edges and openings, regenerating forest, powerline rights-of-way, fencerows, upland thickets, abandoned farms, and shrubby areas along streams, swamps, and ponds. Chats are most abundant in 6- to 9-year-old cottonwood plantations in the Mississippi Alluvial Valley (Twedt and others 1999). However, Annand and Thompson (1997) observed similar abundance across stands subject to alternative forest management prescriptions. In east Texas, density is positively correlated with foliage density at 0 to 3 m, the percentage of saplings that are pine, and the number of shrub species. Densities are negatively affected by increasing vegetation height, percent canopy cover, foliage density at 12 to 15 m, and density of pole trees (Conner and others 1983).

In Missouri, the yellow-breasted chat nests more than 20 m from the edge of large early successional patches characterized by high densities of small stems (Burhans and Thompson 1999). Nest success increases with patch size; territories range from 0.5 to 1.6 ha.

### Model Description

The HSI model for the yellow-breasted chat includes six variables: landform, landcover, successional age class, edge, early successional patch size, and small stem (< 2.5 cm d.b.h.) density.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 157). We directly assigned SI scores to these combinations based on data from Hamel (1992). However, we assumed that shrub-seedling habitats were optimal and that pole stands were nonhabitat. We ignored landform effects in assessing habitat suitability for this species.

Chats prefer to nest more than 20 m from the edge of mature forest (SI2) (Woodward and others 2001). Thus, we used a 3 × 3 pixel window (90 x 90 m) to identify suitable early

**Table 157.—Relationship of landform, landcover type, and successional age class to suitability index scores for yellow-breasted chat habitat**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.000 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Evergreen | 0.333 | 0.667 | 0.500 | 0.000 | 0.000 |
| | Mixed | 0.333 | 0.667 | 0.334 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.667 | 0.334 | 0.000 | 0.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.333 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Deciduous | 0.167 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Evergreen | 0.333 | 0.667 | 0.500 | 0.000 | 0.000 |
| | Mixed | 0.333 | 0.667 | 0.334 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.667 | 0.334 | 0.000 | 0.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Transitional-shrubland | 0.333 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Deciduous | 0.333 | 1.000 | 0.500 | 0.000 | 0.000 |
| | Evergreen | 0.333 | 0.667 | 0.500 | 0.000 | 0.000 |
| | Mixed | 0.333 | 0.667 | 0.334 | 0.000 | 0.000 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.667 | 0.334 | 0.000 | 0.000 |

successional forest sites immediately adjacent to pole or sawtimber successional age class forest. We reduced the suitability of these sites by half (SI score = 0.500; Table 158).

**Table 158.—Influence of edge on suitability index (SI) scores for yellow-breasted chat habitat**

| 3 × 3 pixel window around early successional pixel includes mature forest[a] | SI score |
|---|---|
| Yes | 0.5 |
| No | 1.0 |

[a] Early successional = grass-forb, shrub-seedling, and sapling successional age classes; mature forest = pole or sawtimber successional age classes.

The yellow-breasted chat is associated with large patches of early successional forest (SI3). We aggregated all grass-forb, shrub-seedling, and sapling successional age class sites to calculate patch sizes for this species. We fit a logarithmic function (Fig. 95) to data from Rodewald and Vitz (2005) on the relative abundance of the yellow-breasted chat in early successional patches of various sizes to quantify the relationship between patch size and habitat suitability (Table 159).

This species occupies sites with high small stem densities (SI4). Therefore, we fit a logistic function (Fig. 96) to data from Annand and Thompson (1997) relating the relative density of the yellow-breasted chat to small stem densities (Table 160) to predict the effect of this habitat characteristic on habitat suitability.

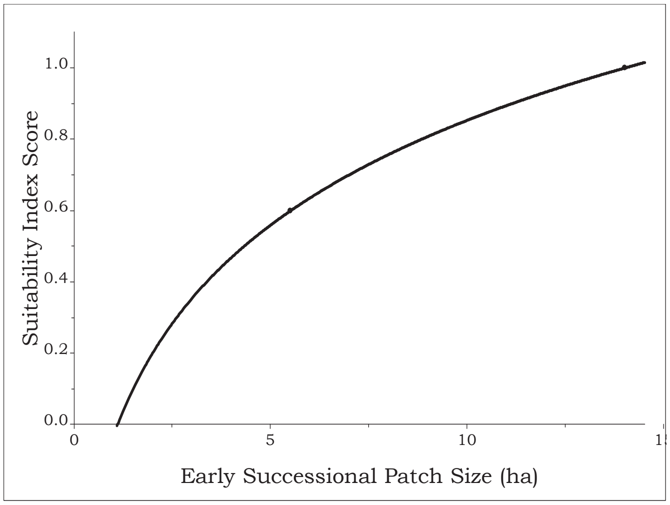

Figure 95.—Relationship between early successional patch size and suitability index (SI) scores for yellow-breasted chat habitat. Equation: SI score = -0.212 + 0.453 * ln(forest patch size).

**Table 159.—Influence of early successional patch size on suitability index (SI) scores for yellow-breasted chat habitat; early successional patches only include grass-forb, shrub-seedling, and sapling successional age classes**

| Early successional patch size (ha)[a] | SI score |
|---|---|
| 6 | 0.6 |
| 14.5 | 1.0 |

[a] Rodewald and Vitz (2005).

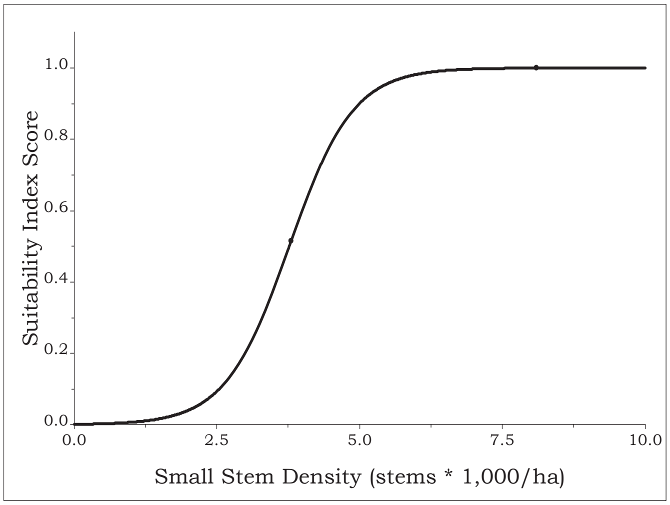

Figure 96.—Relationship between small stem (< 2.5 cm d.b.h.) density (stems * 1000/ha) and suitability index (SI) scores for yellow-breasted chat habitat. Equation: SI score = (1.000 / (1 + (1148216.200 * e $^{-3.689 * (\text{small stem density / 1000})}$))).

**Table 160.—Influence of small stem (< 2.5 cm d.b.h.) density (stems * 1,000/ha) on suitability index (SI) scores for yellow-breasted chat habitat**

| Small stem density[a] | SI score |
|---|---|
| 0.0 | 0.000 |
| 3.8 | 0.516 |
| 8.1 | 1.000 |

[a] Annand and Thompson (1997).

To calculate the overall HSI score for the yellow-breasted chat, we determined the geometric mean of the SI scores for forest structure attributes (SI1 and SI4) and the SI score for landscape composition (SI2 and SI3) separately and then the geometric mean of these values together.

Overall HSI = $((\text{SI1} * \text{SI4})^{0.500} * (\text{SI2} * \text{SI3})^{0.500})^{0.500}$

### Verification and Validation

The yellow-breasted chat was found in all 88 subsections of the CH and WGCP. Spearman rank correlation identified a significant ($P \leq 0.001$) positive relationship ($r_s = 0.40$) between average HSI score and mean BBS route abundance across subsections. The generalized linear model predicting BBS abundance from BCR and HSI for the yellow-breasted chat was significant ($P \leq 0.001$; $R^2 = 0.379$), and the coefficient on the HSI predictor variable was both positive ($\beta = 93.367$) and significantly different from zero ($P \leq 0.001$). Therefore, we considered the HSI model for the yellow-breasted chat both verified and validated (Tirpak and others 2009a).

## Yellow-throated Vireo

Chandler S. Robbins,
Patuxent Bird Identification InfoCenter
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### Status

The yellow-throated vireo (*Vireo flavifrons*) is a neotropical migrant found throughout North America east of the Great Plains. Populations in both the CH and WGCP are stable (Sauer and others 2005) (Table 5). This species is not a Bird of Conservation Concern in either region (Table 1) but is a planning and responsibility species in both the CH (regional combined score = 16) and WGCP (regional combined score = 15). Approximately 20 percent of the continental population breeds in these two BCRs (Panjabi and others 2001).

### Natural History

The yellow-throated vireo breeds along the edges of mature forest stands; its abundance may even decline within forest interiors (Rodewald and James 1996). Appropriate edges include streams, rivers, swamps, and roads. Parks, orchards, and suburban habitats also may be used (Rodewald and James 1996). This species uses both bottomland and upland sites but is restricted to deciduous and mixed-forest habitats. As a forest edge species, it is not area sensitive and may benefit from canopy gaps. However, Robbins and others (1989) observed a positive relationship between the abundance of the yellow-throated vireo and forest cover within a 2-km buffer. Similarly, this bird did not use riparian forests strips that were less than 70 m wide in east Texas (Conner and others 2004). Thus, the yellow-throated vireo prefers canopy gaps within forested landscapes. The key component of its habitat is canopy structure, and this species selects taller trees (> 20 m) than other vireos (James 1976). Robbins and others (1989) also noted a positive relationship between abundance and canopy height. Specific tree species do not affect selection (Gabbe and others 2002).

### Model Description

Our HSI model for the yellow-throated vireo includes six variables: landform, landcover, successional age class, forest patch size, percent forest in the landscape (1-km radius), and canopy cover.

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 161). We directly assigned SI scores to these combinations on the basis of relative rankings of habitat associations for the yellow-throated vireo described in Hamel (1992).

Although a forest edge species, the yellow-throated vireo is affected by forest area (SI2) and the percentage of forest in the landscape (SI3). We fit a logarithmic function (Fig. 97) to data from Blake and Karr (1987) and Kilgo and others (1998) to describe the relationship between forest patch size and habitat suitability (Table 162). Similarly, we used a logistic function to predict habitat suitability from percent forest cover in a 1-km radius landscape (Fig. 98) based on data (Table 163) derived from Donovan and others (1997), who observed differences in predator and brood parasite communities among highly fragmented (< 15

**Table 161.—Relationship of landform, landcover type, and successional age class to SI scores for yellow-throated vireo habitat**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.333 | 0.667 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.333 | 0.667 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.167 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.333 | 0.667 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.417 | 0.834 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.250 | 0.500 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.250 | 0.500 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.167 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.250 | 0.500 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.500 | 1.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.334 | 0.667 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.334 | 0.667 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.167 | 0.333 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.334 | 0.667 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.500 | 1.000 |

percent), moderately fragmented (45 to 50 percent), and lightly fragmented (> 90 percent forest) landscapes. We assumed that the midpoints between these classes (30 and 70 percent forest) defined the specific cutoffs for poor (SI score ≤ 0.10) and excellent (SI score ≥ 0.90) habitat, respectively.

The affinity of the yellow-throated vireo for canopy gaps led us to incorporate canopy cover in the HSI model for this species (SI4). We fit a smoothed quadratic function (Fig. 99) to data from Kahl and others (1985) (Table 164) on the relative density of this species at varying canopy closures, and assumed that Kahl's optimal designation of canopy cover (80 to 90 percent) was associated with maximum SI scores. Further, we assumed that habitat suitability declined symmetrically as canopy cover departed from this optimum.

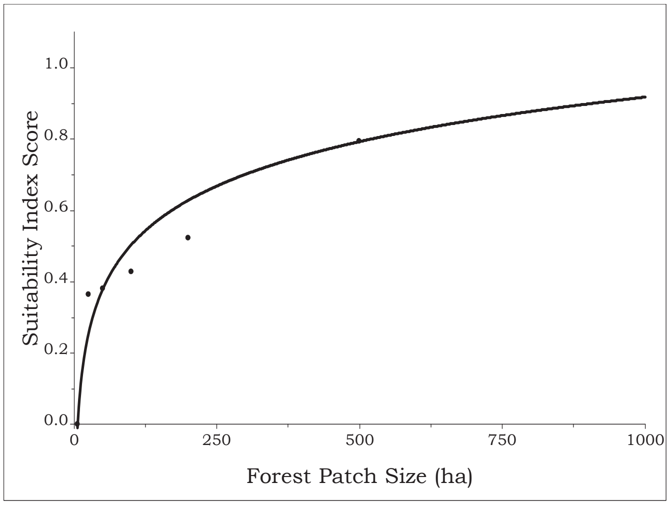

Figure 97.—Relationship between forest patch size and suitability index (SI) scores for yellow-throated vireo habitat. Equation: SI score = 0.180 * ln(forest patch size) – 0.323.

**Table 162.—Influence of forest patch size on suitability index (SI) scores for yellow-throated vireo habitat**

| Forest patch size (ha) | SI score |
|---|---|
| 6.5[a] | 0.000 |
| 25[b] | 0.365 |
| 50[b] | 0.381 |
| 100[b] | 0.429 |
| 200[b] | 0.524 |
| 500[b] | 0.794 |
| 1000[b] | 1.000 |

[a] Blake and Karr (1987).

[b] Kilgo and others (1998).

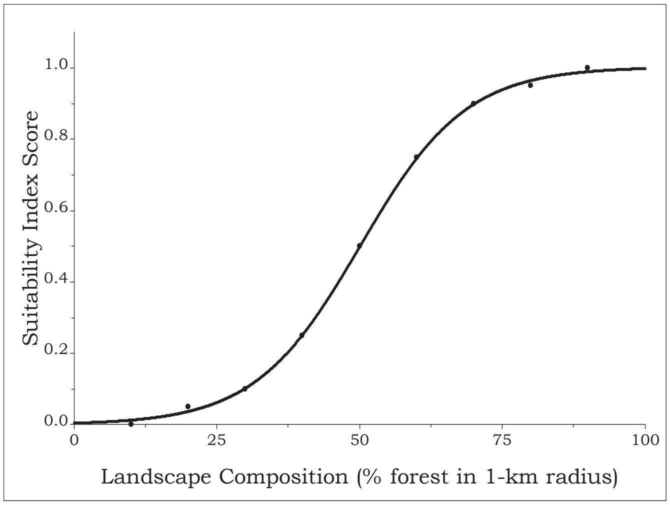

Figure 98.—Relationship between landscape composition and suitability index (SI) scores for yellow-throated vireo habitat. Equation: SI score = 1.005 / (1.000 + (221.816 * $e^{-0.108 * (\text{landscape composition})}$)).

**Table 163.—Relationship between landscape composition (percent forest in 1-km radius) and suitability index (SI) scores for yellow-throated vireo habitat**

| Landscape composition | SI score |
|---|---|
| 0[a] | 0.00 |
| 10[a] | 0.00 |
| 20[a] | 0.05 |
| 30[b] | 0.10 |
| 40[a] | 0.25 |
| 50[b] | 0.50 |
| 60[a] | 0.75 |
| 70[b] | 0.90 |
| 80[a] | 0.95 |
| 90[b] | 1.00 |
| 100[a] | 1.00 |

[a] Assumed value.

[b] Donovan and others (1997).

To calculate the overall HSI score, we determined the geometric mean of SI scores for forest structure (SI1 and SI4) and landscape composition attributes (SI2 and SI3) separately and then the geometric mean of these means together.

Overall HSI = $((\text{SI1} * \text{SI4})^{0.500} * (\text{SI2} * \text{SI3})^{0.500})^{0.500}$

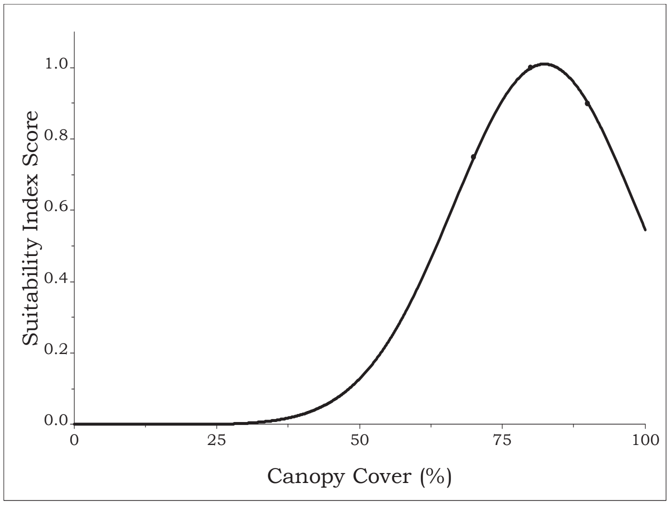

Figure 99.—Relationship between canopy cover and suitability index (SI) scores for yellow-throated vireo habitat. Equation: SI score = 1.011 * e $^{(0-((\text{canopy cover} - 82.319)\text{\^{}}2 / 508.869))}$.

**Table 164.—Influence of canopy cover (percent) on suitability index (SI) scores for yellow-throated vireo habitat**

| Canopy cover (percent) | SI score |
|---|---|
| 0[a] | 0.00 |
| 70[b] | 0.75 |
| 80[b] | 1.00 |
| 90[a] | 0.90 |

[a] Assumed value.
[b] Kahl and others (1985).

## Verification and Validation

The yellow-throated vireo was found in all 88 subsections of the CH and WGCP. Spearman rank correlation on average HSI score and mean BBS route abundance per subsection identified a significant ($P \leq 0.001$) positive association ($r_s = 0.51$) between these two variables. The generalized linear model predicting BBS abundance from BCR and HSI for the yellow-throated vireo was significant ($P = 0.002$; $R^2 = 0.133$), and the coefficient on the HSI predictor variable was both positive ($\beta = 2.811$) and significantly different from zero ($P \leq 0.001$). Therefore, we considered the HSI model for the yellow-throated vireo both verified and validated (Tirpak and others 2009a).

## Yellow-throated Warbler

Deanna K. Dawson, Patuxent Bird Identification InfoCenter
Photo used with permission

### Status

The yellow-throated warbler (*Dendroica dominica*) is a neotropical migrant that breeds in the southeastern United States and reaches its highest densities in the Ohio River Valley. This species has remained relatively stable in the WGCP over the past 40 years but has increased considerably in the CH (3.8 percent per year since 1967; Table 5). The yellow-throated warbler is not a Bird of Conservation Concern in either BCR but is a planning and responsibility species in the CH (regional combined score = 15; Table 1).

### Natural History

The yellow-throated warbler breeds in two distinct habitat types: mature bottomland hardwood forest and dry, upland oak-pine forest (Hall 1996). It is more common in the former. This species shows a strong affinity for cypress along the Coastal Plains, but prefers sycamore along inland rivers (Hall 1996, Gabbe and others 2002). Where Spanish moss is found, it is used for both foraging and nesting (Hall 1996). Elsewhere, the warbler forages by creeping along limbs and probing leaf clusters and pinecones. This bird is both an interior and edge species and may occupy woodlots as small as 6 ha (Blake and Karr 1987). Robbins and others (1989) associated this species with large tree (> 38 cm d.b.h.) density, forest in a 2-km buffer, and coniferous canopy cover.

### Model Description

Our HSI model for the yellow-throated warbler includes six variables: landform, landcover, successional age class, large tree (> 50 cm d.b.h.) density, distance to water, and percent forest in the landscape (1-km radius).

The first suitability function combines landform, landcover, and successional age class into a single matrix (SI1) that defines unique combinations of these classes (Table 165). We directly assigned SI scores to these combinations on the basis of habitat associations outlined by Hamel (1992) for the yellow-throated warbler in the Southeast.

We also incorporated large tree density (SI2) into the HSI model for the yellow-throated warbler because of its affinity for nesting and foraging in large trees (Hamel 1992, Robbins and others 1989). Lacking data points from the literature to fit a curve, we assumed that SI scores were logistically related to large tree density up to 50 trees per ha and remained optimal above this threshold (Fig. 100, Table 166).

The yellow-throated warbler typically nests near water (Hall 1996, Hamel 1992). Thus, we included distance to water (SI3) in the HSI model. We assumed that sites closer to water had a higher suitability. Lacking quantitative data on the potential effect of water on habitat suitability, we assumed that the size of the yellow-throated warbler's territory is similar to that of the Acadian flycatcher but that the warbler is not as dependent on water as the

**Table 165.—Relationship of landform, landcover type, and successional age class to suitability index (SI) scores for yellow-throated warbler habitat; values in parentheses apply to West Gulf Coastal Plain/Ouachitas**

| | | Successional age class | | | | |
|---|---|---|---|---|---|---|
| Landform | Landcover type | Grass-forb | Shrub-seedling | Sapling | Pole | Saw |
| Floodplain-valley | Low-density residential | 0.000 | 0.000 | 0.000 | 0.250 | 0.500 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.250 | 0.500 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.333 | 0.667 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.333 | 0.667 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.834 | 1.000 |
| Terrace-mesic | Low-density residential | 0.000 | 0.000 | 0.000 | 0.167 | 0.167 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.167 | 0.167 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.333 | 0.667 |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.333 | 0.667 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.500 | 1.000 |
| Xeric-ridge | Low-density residential | 0.000 | 0.000 | 0.000 | 0.167 | 0.333 |
| | Transitional-shrubland | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Deciduous | 0.000 | 0.000 | 0.000 | 0.167 | 0.333 |
| | Evergreen | 0.000 | 0.000 | 0.000 | 0.333<br>(0.167) | 0.667<br>(0.334) |
| | Mixed | 0.000 | 0.000 | 0.000 | 0.333 | 0.667 |
| | Orchard-vineyard | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| | Woody wetlands | 0.000 | 0.000 | 0.000 | 0.500 | 1.000 |

flycatcher. Therefore, we assumed that all sites less than 100 m from water were optimal but reduced SI more slowly for the yellow-throated warbler than the Acadian flycatcher as distance to water increased (Fig. 101; Table 167).

The yellow-throated warbler responds to the percentage of forest in the landscape (SI4). To capture this relationship, we fit a logistic function (Fig. 102) to data (Table 168) derived from Donovan and others (1997), who observed differences in predator and brood parasite communities among highly fragmented (< 15 percent), moderately fragmented (45 to 50 percent), and lightly fragmented (> 90 percent forest) landscapes. We assumed that the midpoints between these classes (30 and 70 percent forest) defined the specific cutoffs for poor (SI score ≤ 0.10) and excellent (SI score ≥ 0.90) habitat, respectively.

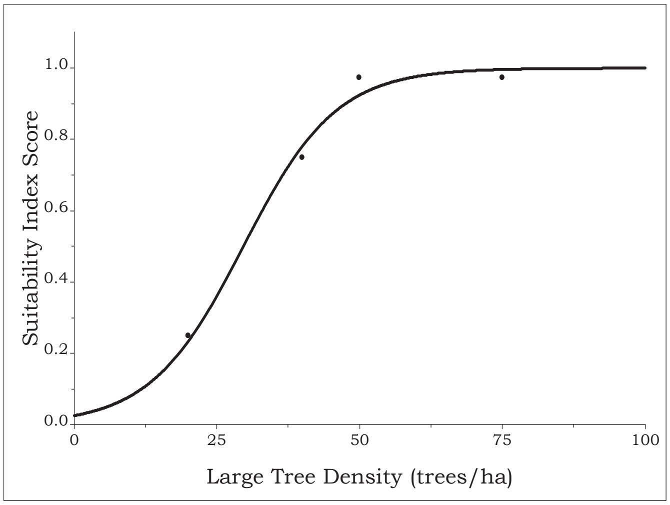

Figure 100.—Relationship between large tree (> 50 cm d.b.h.) density and suitability index (SI) scores for yellow-throated warbler habitat. Equation: SI score = 1.000 / (1.0000 + (38.185 * $e^{-0.123 * \text{large tree density}}$)).

**Table 166.—Influence of large tree (> 50 cm d.b.h.) density (trees/ha) on suitability index (SI) scores for yellow-throated warbler habitat**

| Large tree density[a] | SI score |
|---|---|
| 0 | 0.00 |
| 20 | 0.25 |
| 40 | 0.75 |
| 50 | 1.00 |
| 75 | 1.00 |

[a] Assumed value.

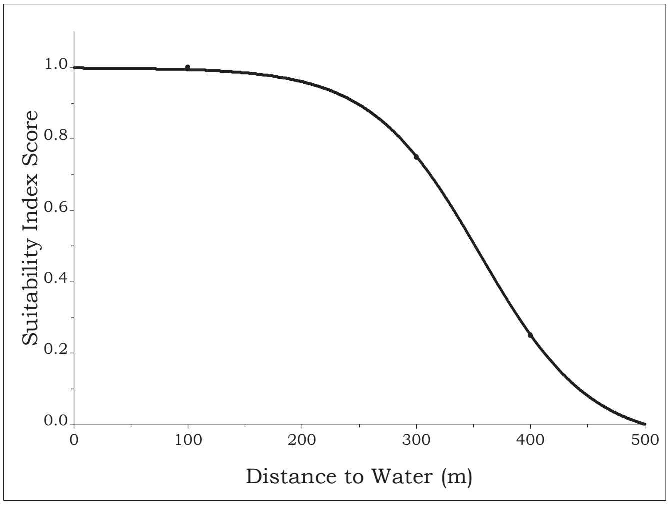

Figure 101.—Relationship between distance to water and suitability index (SI) scores for yellow-throated warbler habitat. Equation: SI score = 1 - (1.050 / (1 + (1661.322 * $e^{-0.021 * \text{distance to water}}$))).

**Table 167.—Relationship between distance to water and suitability index (SI) scores for yellow-throated warbler habitat**

| Distance to water (m)[a] | SI score |
|---|---|
| 100[b] | 1.00 |
| 300[b] | 0.75 |
| 400[b] | 0.25 |
| 500[b] | 0.00 |

[a] Water defined as NHD streams or NLCD water, woody wetlands, and emergent herbaceous wetlands classes.
[b] Assumed value.

To calculate the overall HSI score, we determined the geometric mean of SI scores for forest structure (SI1 and SI2) and landscape composition attributes (SI3 and SI4) separately and then the geometric mean of these means together.

Overall HSI = $((SI1 * SI2)^{0.500} * (SI3 * SI4)^{0.500})^{0.500}$

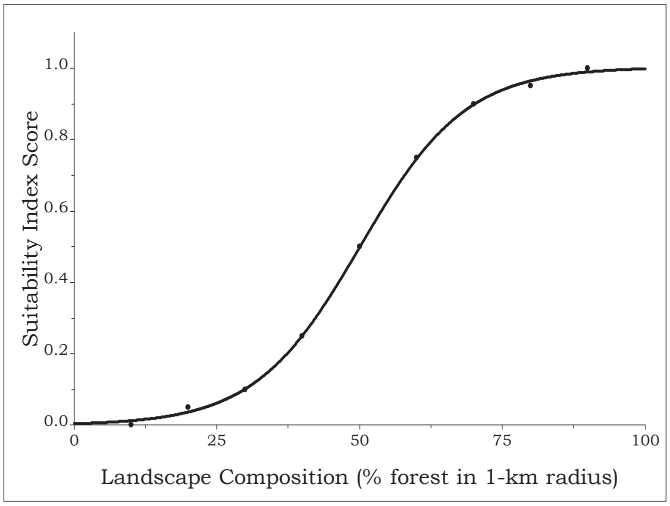

Figure 102.—Relationship between landscape composition and suitability index (SI) scores for yellow-throated warbler habitat. Equation: SI score= 1.005 / (1.000 + (221.816 * e $^{-0.108 * \text{(landscape composition)}}$)).

**Table 168.—Relationship between landscape composition (percent forest in 1-km radius) and suitability index (SI) scores for yellow-throated warbler habitat**

| Landscape composition | SI score |
|---|---|
| 0[a] | 0.00 |
| 10[a] | 0.00 |
| 20[a] | 0.05 |
| 30[b] | 0.10 |
| 40[a] | 0.25 |
| 50[b] | 0.50 |
| 60[a] | 0.75 |
| 70[b] | 0.90 |
| 80[a] | 0.95 |
| 90[b] | 1.00 |
| 100[a] | 1.00 |

[a] Assumed value.

[b] Donovan and others (1997).

## Verification and Validation

The yellow-throated warbler was found in 87 of the 88 subsections within the CH and WGCP. Spearman rank correlation on average HSI score and mean BBS route abundance identified a significant ($P \leq 0.001$) positive association ($r_s = 0.48$) between these two variables within subsections where this species was detected. The generalized linear model predicting BBS abundance from BCR and HSI for the yellow-throated warbler was significant ($P = 0.003$; $R^2 = 0.125$), and the coefficient on the HSI predictor variable was both positive ($\beta = 2.870$) and significantly different from zero ($P = 0.020$). Therefore, we considered the HSI model for the yellow-throated warbler both verified and validated (Tirpak and others 2009a).

## CURRENT MODEL USE AND FUTURE DIRECTIONS

For species with verified and validated models, we developed geospatial datasets that summarize the habitat suitability and estimated population size of these species within each subsection for two periods (1992 and 2001). These datasets are being used to assess changes in habitats through time and identify which model variables are associated with these changes. We also are using these datasets as conservation design tools to identify the specific location and type of management practice that may most effectively increase the habitat quality and population size of target species. Population estimates explicitly tied to habitat suitability are allowing the refinement of landbird population objectives and spatial depiction of these objectives at the ecological subsection scale. We are developing a decision-support tool based on these model outputs that will estimate the magnitude of management that may be required to achieve population objectives for a particular species and will assess the simultaneous impacts of different management options on populations of multiple species.

With conservation informed by these models in both the CH and WGCP, these models are informing the status at the continental scale of species with a significant portion of their populations in these BCRs (e.g., Kentucky warbler; Panjabi and others 2005). Adoption and application of these models in other BCRs (the East Gulf Coastal Plain Joint Venture references the use of these models in its Implementation Plan [East Gulf Coastal Plain Joint Venture 2008]) may provide a framework for assessing the status of additional species at the continental scale. However, the use of these models outside the CH and WGCP will require careful scrutiny and additional testing to ensure that the habitat associations remain valid as differences in forest types among regions (particularly outside the Southeast) likely will affect the SI scores in the landform, forest type, and successional age class matrix derived from Hamel (1992).

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Habitat Suitability Index (HSI) models were developed to assess habitat quality for 40 priority bird species in the Central Hardwoods and West Gulf Coastal Plain/ Ouachitas Bird Conservation Regions. The models incorporated both site and landscape environmental variables from one of six nationally consistent datasets. Potential habitat was first defined from unique landform, landcover, and successional age class combinations. Species-specific environmental variables identified from the literature were used to refine initial habitat estimates. Models were verified by comparing subsection-level HSI scores and Breeding Bird Survey (BBS) abundance via Spearman rank correlation. Generalized linear models that predicted BBS abundance as a function of HSI were used to validate models.

KEY WORDS: Conservation planning, ecoregion, forest, Forest Inventory and Analysis, National Landcover Dataset, validation.

*Capitalizing on the strengths of existing science capacity in the Northeast and Midwest to attain a more integrated, cohesive, landscape-scale research program*

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