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Using urban forest assessment tools to model bird habitat potential

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Using urban forest assessment tools to model bird habitat potential

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HIGHLIGHTS

- The i-Tree wildlife tool assesses the bird habitat potential within the urban forest.
- The i-Tree wildlife tool evaluates habitat improvement plans.
- The i-Tree wildlife tool provides detailed information of habitat requirements.

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ABSTRACT

The alteration of forest cover and the replacement of native vegetation with buildings, roads, exotic vegetation, and other urban features pose one of the greatest threats to global biodiversity. As more land becomes slated for urban development, identifying effective urban forest wildlife management tools becomes paramount to ensure the urban forest provides habitat to sustain bird and other wildlife populations. The primary goal of this study was to integrate wildlife suitability indices to an existing national urban forest assessment tool, i-Tree. We quantified available habitat characteristics of urban forests for ten northeastern U.S. cities, and summarized bird habitat relationships from the literature in terms of variables that were represented in the i-Tree datasets. With these data, we generated habitat suitability equations for nine bird species representing a range of life history traits and conservation status that predicts the habitat suitability based on i-Tree data. We applied these equations to the urban forest datasets to calculate the overall habitat suitability for each city and the habitat suitability for different types of land-use (e.g., residential, commercial, parkland) for each bird species. The proposed habitat models will help guide wildlife managers, urban planners, and landscape designers who require specific information such as desirable habitat conditions within an urban management project to help improve the suitability of urban forests for birds.

1. Introduction

The modification and destruction of wildlife habitat within urban areas via the replacement of forest cover and native vegetation with lawns, buildings, roads, and other impervious surfaces poses one of the greatest threats to bird populations on a global scale (Czech, Krausman, & Devers, 2000). Replacing native vegetation with ornamentals is one of the forms that habitat alterations take in the urban environment, and these esthetically pleasing landscapes are often at odds with ecological function (Lerman, Turner, & Bang, 2012). Thus, wildlife management tools aimed at assessing and improving urban habitat have an important role to play in reversing the loss of urban biodiversity.

Urban and community areas in the conterminous United States on average have 35% tree cover (Nowak & Greenfield, 2012), though the resulting urban landscape is a mix of contiguous (e.g., forest stands in parks or vacant areas) and fragmented (e.g., isolated trees along streets and in private yards) cover. Over the next 50 years, it is estimated that 118,300 km^2 of forested lands in the US will be consumed by urbanization (Nowak & Walton, 2005). Nonetheless, the urban forest provides essential ecosystem services that sustain environmental quality and human health (Nowak & Walton,

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2005). In particular, trees and other urban vegetation help mitigate the urban heat island effect through evapotranspiration and by providing shade, and they reduce air pollution through carbon sequestration (Akbari, Pomerantz, & Taha, 2001). Furthermore, the urban forest provides wildlife habitat resources including food, and nest and roosting sites for birds, mammals, and insects. And finally, the urban forest provides opportunities for urbanites to connect with the natural world (Miller, 2005). Currently we lack methods for a rapid assessment of the habitat potential of the urban forest (Shanahan, Possingham, & Martin, 2011). Therefore designing effective urban habitat assessment tools that can assist with the reconciliation between urban development and wildlife habitat becomes paramount to ensure that conservation efforts and plans for enhancing and protecting the urban forest will lead to sustainable bird and other desirable wildlife populations.

Few North American federal and Non-governmental Organization (NGO) programs have targeted improvement plans in urban habitats. The North American Landbird Conservation Plan (NALCP; Rich et al., 2004) aims to create and conserve landscapes that sustain bird populations. The NALCP calls for a thorough examination into how birds respond to and tolerate different land uses, including suburban areas, and recognizes the imminent threat of urbanization to most of the primary bird habitats in North America. Other than encouraging bird-friendly urban planning, the NALCP primarily characterizes urban areas as a threat to bird populations on a national scale without acknowledging the many opportunities for promoting conservation initiatives in urban and suburban landscapes (Goddard, Dougill, & Benton, 2010). The U.S. Fish and Wildlife Service's Urban Bird Treaty program (U.S. Fish and Wildlife Service, 2012) provides competitive challenge grants to individual cities for promoting education, hazard reduction, and habitat improvement projects aimed at supporting native urban bird populations. The National Wildlife Federation and the National Audubon Society have programs aimed at creating and certifying wildlife habitats in residential gardens and schoolyards with their respective Certified Wildlife Habitat and Healthy Yards programs. Although effective and innovative at the site level, these programs do not include management or monitoring programs for urban bird populations at regional scales. Recently Partners in Flight (PIF; an international cooperative effort that partners federal, state and local government agencies, NGOs, academia, and private landowners to conserve species at risk) recognized the extent of urban areas and the negative impact of urbanization on bird populations (Berlanga et al., 2010), though currently, PIF does not focus efforts toward conserving or enhancing urban habitats (Watts, 1999).

Scientists have studied urban bird populations since the 1970s (e.g., Emlen, 1974), however, our understanding of urban habitat and bird relationships trails behind that of habitat relationships in wildlands, thus hindering effective regional conservation plans aimed at improving bird habitat within the urban forest. Studying bird habitat relationships date back to the early 1900s (e.g., Adams, 1935; Grinnell, 1917; Lack, 1933). This research and other seminal works provided the foundation for understanding the habitat requirements for sustaining bird populations and have guided conservation planning, such as the NALCP (Fitzgerald et al., 2009). To date, the majority of urban bird studies conduct a bird monitoring protocol to document distribution patterns, measure habitat features at local and landscape scales, and design statistical models to identify the habitat features that relate to and influence patterns of bird abundance (Chace & Walsh, 2006). In addition, many urban bird studies correlate bird distribution with habitat features measured along an urban to rural gradient, within different land-use categories, or between urban and wildland sites (Beissinger & Osborne, 1982; Blair, 1996; Clergeau, Savard, Mennechez, & Falardeau, 1998; Croci, Butet, & Clergeau, 2008; Crooks, Suarez, & Bolger, 2004; DeGraaf & Wentworth, 1986; Emlen, 1974; Gering & Blair, 1999; Lerman & Warren, 2011; Melles, 2005). Additional variables identified as important in influencing urban bird populations include household density, human activities, and socio-economics (Fernandez-Juricic, 2000; Kinzig, Warren, Martin, Hope, & Katti, 2005; Lerman & Warren, 2011; Strohbach, Haase, & Kabisch, 2009).

Although these and other studies provide a solid foundation for understanding how birds respond to conditions within a particular city, they lack a means for non-specialists to apply these findings to conservation planning and management. In an effort to provide such tools, Tirpak and colleagues and Jones-Farrand and colleagues modeled how patch and landscape habitat features influence suitability for birds at an ecoregional scale (Tirpak, Jones-Farrand, Thompson, Twedt, & Uihlein, 2009; Jones-Farrand et al., 2011). Using the USDA Forest Service national forest census program Forest Inventory and Analysis (FIA) datasets, they described the forest structure and composition in the central and south-central U.S. and constructed Habitat Suitability Index (HSI) models that quantitatively relate forest characteristics to the abundance of forty bird species of conservation concern. They validated the models with Breeding Bird Survey data by testing whether the predicted suitability of landscapes based on the FIA and other data accorded with presence and relative abundance of a particular species (Tirpak, Jones-Farrand, Thompson, Twedt, Baxter, et al., 2009). These models have tremendous management potential in that they can assess the suitability at an ecoregional scale by leveraging existing forest and bird monitoring programs. Further, they assess habitat in terms of manageable characteristics such that they can be used to guide management prescriptions and predict the response of birds to various management scenarios.

Here we introduce the approach of integrating two existing bird habitat models (e.g., Tirpak, Jones-Farrand, Thompson, Twedt, Baxter, et al., 2009) and developing seven new models using the same model building procedure, and integrate these models into an urban forest assessment tool to evaluate the potential of the urban forest for supporting breeding bird populations, while also providing a platform for generating habitat improvement plans. This study aims to describe and validate the habitat models, and to demonstrate their applicability for improving urban bird diversity. Specifically we (1) identified the vegetation composition, configuration, and landscape features associated with the presence of a suite of representative bird species based on an extensive literature review, (2) quantified the characteristics of urban forests in ten northeastern cities using datasets from the i-Tree urban forest assessment program (Nowak et al., 2008), (3) modeled the habitat suitability for the representative bird species in urban forest monitoring plots, validated the models, and compared habitat suitability among ten cities and different land uses, and (4) tested whether habitat suitability changed over time for two cities for which we had habitat data for two points in time.

2. Methods

2.1. Study area

This study assesses the habitat potential for ten northeastern U.S. cities (Baltimore, MD, Boston, MA, Jersey City, NJ, Moorestown, NJ, New York, NY, Philadelphia, PA, Scranton, PA, Syracuse, NY, Washington D.C., and Woodbridge, NJ). These cities were selected because they had available urban forest data from i-Tree, and had a wide range of population sizes (19,000 – 8.4 million). Cities ranged from small municipalities such as Moorestown, NJ to large metropolitan areas such as Boston and Philadelphia, and thus were representative of urban areas in the region.

Table 1
Bird species list with associated life history traits, conservation status, and eBird frequencies (mean, minimum and maximum) included in the i-Tree wildlife habitat models. Forage and nest guilds include primary foraging and nesting locations. A conservation status of PIF indicates a Partners In Flight species of conservation concern.

Species	Summer frequency (ranges)	Forage guild	Nest guild	Conservation
American Robin	0.64 (0.50–0.79)	Lower canopy/ground	Tree branch	Flagship
Baltimore Oriole	0.25 (0.16–0.39)	Lower/upper canopy	Tree twig	PIF
Black-capped Chickadee	0.24 (0.03–0.56)	Lower canopy	Tree cavity	Flagship
Carolina Chickadee	0.28 (0.22–0.37)	Lower canopy	Tree cavity	PIF
European Starling	0.53 (0.38–0.70)	Ground	Buildings/cavities	Invasive
Northern Cardinal	0.49 (0.29–0.65)	Ground	Shrubs	Flagship
Red-bellied Woodpecker	0.19 (0.03–0.33)	Bark	Tree cavity	Flagship
Scarlet Tanager	0.08 (0.01–0.16)	Upper canopy	Tree twig	PIF
Wood Thrush	0.14 (0.03–0.25)	Ground	Tree branch	PIF

2.2. Bird species selection

In order to identify candidate bird species for this study, we first generated bird lists and average frequencies for all species recorded during the breeding season (mid-May through June in the northeast region) from 1990 to 2000, in the ten cities (i.e., their associated counties) using the Cornell Lab of Ornithology eBird database (eBird, 2012). The eBird database includes lists of birds seen during outings by amateur participants, and vetted by experts, and then uploaded with locality data, to an accessible interactive web-platform. Frequencies represented the percentage of submitted eBird checklists that record a particular species. We then identified the species recorded in all ten cities and calculated the mean, minimum and maximum frequency for each species. A total of 204 species were recorded in all ten cities, though only 57 species had frequencies >0.05. Species with few records (i.e., frequencies) are often not accurately placed in ecological space and hence we did not include species with frequencies <0.05 (McCune & Grace, 2002). Furthermore, the majority of species with low frequencies were forest interior species, species prone to local extinction within small and isolated forest fragments (Sherry & Holmes, 1985), and unlikely to penetrate the urban forest (Blair, 1996).

The urban forest could be important for birds in a number of ways. For instance, some forest interior species might penetrate the urban matrix when large tracts of forest exist. These rare species might be of particular concern because their populations might be vulnerable (Miller & Hobbs, 2002), and therefore we included species with differing levels of reporting frequencies (>0.05 frequency). The characteristic strata or substrate a bird uses for foraging or nesting could indicate the presence of resources needed by other species (Simberloff & Dayan, 1991), so we included species from a diversity of foraging and nesting guilds. Finally, species differed in their conservation significance. We included species recognized as high conservation priority, invasive or important for cultural reasons. Four of the selected species had a Partners in Flight (PIF) designation which ranks a species' conservation vulnerability based on "global measures, threats to breeding populations, area importance, and population trend for specific physiographic areas", and conservation initiatives and plans are directed toward species with high PIF scores (Rich et al., 2004). Invasive species included exotic birds that exploit the urban landscape (Blair, 1996). Urban flagship species were birds that urbanites recognize and embrace, following Caro and O'Doherty (1999). We ensured the species selected represented different foraging and nesting guilds with a focus on guilds reliant on forests (DeGraaf, Tilghman, & Anderson, 1985). Our final list included nine bird species with varying abundances, life history traits, and conservation status (Table 1).

2.3. i-Tree data

We used data from the above-mentioned 10 northeastern cities that were analyzed using the i-Tree model (www.itreetools.org; formerly known as the Urban Forest Effects [UFORE] model) for our habitat modeling. The i-Tree program is a free suite of tools developed by the US Forest Service to assess the ecosystem services and values provided by the urban forest. This program is designed to aid in the understanding and management of urban forests to help sustain environmental quality and human health in cities across the nation. The tool integrates local field data (e.g., species, tree height, canopy percentage) from either complete inventories or plot-based samples of trees with local air pollution and meteorological data to quantify forest structure and calculate the ecosystem services and values provided by the urban forest (Nowak et al., 2008). Data from i-Tree has provided information on the value of urban trees and their capacity to store carbon, mitigate energy costs, and remove air pollution (e.g., Nowak, Crane, & Stevens, 2006; Nowak, Greenfield, Hoehn, & Lapoint, 2013; Nowak, Hirabayshi, Bodine, & Hoehn, 2013). Information gathered via i-Tree has helped scientists to link urban forest management with environmental quality, and has assisted managers with planning for the future (Driscoll et al., 2012). Currently, the tool lacks the capacity to assess the habitat potential, an additional ecosystem service of the urban forest.

Each city included about 200 randomly selected plots (0.04 ha) located among all land-use categories (e.g., residential, commercial, parkland, and agricultural). Data collected at each plot included tree characteristics, percent cover of buildings, grass, shrubs and trees, the land use, and land cover. For each tree (woody plants with a minimum diameter of 2.54 cm at 1.4 m) numerous variables were collected including tree size, height, and condition (Table 2).

Table 2
List of i-Tree variables included in the i-Tree wildlife habitat models.

Variable	Description
PLOT ID	i-Tree plot identification
LANDUSE	Land-use category for each i-Tree plot
%BLDG	Percent of plot (0.04 ha) with land cover classification of building
%GRASS M	Percent of plot (0.04 ha) with land cover classification of lawn (maintained)
%SHRB	Percent of plot (0.04 ha) with shrub cover
%TREE	Percent of plot (0.04 ha) covered by tree canopy
TR_DENS_ALL	Number of all trees within plot (0.04 ha)
SAP DENS	Number of saplings (<10 cm dbh) within plot (0.04 ha)
23cm_DENS	Number of trees > 23 cm dbh within plot (0.04 ha)
DEAD_DENS	Number of trees within plot (0.04 ha) with fair, poor, dying, dead classification
BA_6 cm	Basal area of trees greater than 6 cm dbh per ha
MEAN_TOT HT_m	Mean tree height (m) per plot (0.04 ha)
FOR_AREA[a]	Amount of contiguous forest area (ha) surrounding i-Tree plot
FOR_1KM[a]	Percent forest land cover within 1 km of i-Tree plot

[a] These variables not collected using i-Tree but will be analyzed using plot location, forest cover maps and GIS analyses.

2.4. Bird habitat models

We conducted extensive literature reviews for each bird species using Web of Science and other databases as well as the literature-cited sections of papers. We identified habitat variables that were found to affect a species' abundance (Jones-Farrand et al., 2011) and also corresponded to measurements in the i-Tree datasets. Although i-Tree data did not always align with habitat variables representative of a particular species, we were able to extract this information from i-Tree and include these important local habitat variables. For example, basal area, a common forestry measurement, was listed in a number of publications describing habitat relationships but was not part of the i-Tree database. Thus we calculated the basal area based on the i-Tree data, and included this variable in two of our models. Similarly with dead wood, an important resource for cavity-nesting species, we extracted the tree condition data from i-Tree and assumed that trees with a rating of fair, poor, dying or dead had dead wood present. We assigned suitability index (SI) scores for each species, for each metric. The SI ranged between 0 and 1 whereby a score of 0 indicated unsuitable habitat conditions (i.e., strong likelihood the species not present) whereas a score of 1 indicated the habitat conditions have a strong likelihood of supporting the species. Often, published data consisted of a single mean value for a habitat feature (e.g., percent canopy cover) when the species was present, and we used this data point when building the models. In instances when published data were scant or not available, we estimated values by supplementing with iterative values which improved the predictability of our habitat models (Tirpak, Jones-Farrand, Thompson, Twedt, & Uihlein, 2009). These and the iterative values mentioned above were reviewed by a panel of experts and revised according to recommendations (Tirpak, Jones-Farrand, Thompson, Twedt, & Uihlein, 2009). Each habitat variable per species included at least three data points. We used CurveExpert Professional software (http://www.curveexpert.net/) to generate parameters for mathematical equations to predict the probability of a species occurrence for each habitat variable (e.g., percent canopy cover) based on the value of that variable. We selected the equation with the best fit to the data (r^2). We identified between two and five habitat variables that were associated with each species, and generated mathematical equations for each habitat variable. We then calculated the geometric mean for these two to five habitat variables used for each species for a final SI score for each plot. This assumes that each variable had equal weight in the model (Jones-Farrand et al., 2011).

These habitat models have various assumptions and limitations associated with their use. First, relying on expert opinion on the estimated values might have introduced observer bias (Jones-Farrand et al., 2011). However, we solicited opinions from at least three different wildlife biologists intimately familiar with our targeted species. Furthermore, we valued expert opinion and have confidence that the inclusion of the estimated values were more informative than having models without these values (Beaudry et al., 2010). We assumed the species were limited in their distribution by the habitat variables selected for the models, and the variables measured in i-Tree represented the suite of habitat variables a particular species used in the selection process (Jones-Farrand et al., 2011). We assumed that behavioral interactions (e.g., inter and intra-specific competition) were not the driving force birds used for selecting habitat (Sherry & Holmes, 1985). We assumed the models performed equally within the different land-uses, for generalist and specialist bird species, and that we built the models based on complete information on habitat relationships. In addition, since the majority of published habitat relationship studies were conducted in wildlands (i.e., not in urban land-uses), we assumed these relationships were applicable to urban landscapes (Beaudry et al., 2010; Roloff & Kernohan, 1999). And finally, the habitat models do not fully account for landscape variables that might indicate the permeability and connectivity throughout the urban landscape, essential factors for dispersal (Beaudry et al., 2010). We included the full description of habitat associations and subsequent models for the red-bellied woodpecker (*Melanerpes carolinus*) to illustrate the habitat model building process. See the online supplementary material for the remaining species accounts and models.

2.5. Validating the models

To test the validity of our habitat models, we used bird monitoring data from 82 sites located at the Baltimore Ecosystem Study Long-Term Ecological Research (BES LTER) project. To the best of our knowledge, Baltimore was the only city in the northeast with an extensive bird monitoring program. In addition, the bird monitoring sites coincided with the i-Tree collection sites and thus enabled us to directly test how the habitat models predicted species presence by comparing the HSI with the presence of a particular species. Each site was visited two times per year (2002, 2004–2007) during the breeding season (mid May to July) by a trained observer. Visits occurred between sunrise and 09:30, and all species heard and seen during the 5-min count were recorded (Nilon, Warren, & Wolf, 2011). Using the point count data, we calculated a mean abundance and categorized each species as present or absent at each i-Tree location. Five of the nine species were recorded at the BES LTER project: American robin (*Turdus migratorius*), Carolina chickadee (*Poecile carolinensis*), European starling (*Sturnus vulgaris*), northern cardinal (*Cardinalis cardinalis*), and red-bellied woodpecker. We compared the HSI scores with the BES LTER bird abundance data using Spearman Rank correlations. We assessed model sensitivity by removing one habitat variable at a time, and recalculated the HSI score to test whether the omission of the said variable altered the predictability of the model. For example, the red-bellied woodpecker model included four habitat variables: the number of large trees, basal area, percent canopy cover and dead wood density. To test whether the model was sensitive to the number of large trees, we generated a new HSI score by calculating the geometric mean of the three other habitat variables and then compared the new HSI score with the BES LTER bird abundance data using Spearman Rank correlations. Discrepancies between the two analyses (i.e., significant with all variables yet not significant with the omitted variable) suggested the omitted habitat variable had a greater influence to the model. Black-capped chickadee (*Poecile atricapillus*) range does not include Baltimore though we used Carolina chickadee model for validation. Tirpak, Jones-Farrand, Thompson, Twedt, and Uihlein (2009) used Breeding Bird Survey (BBS) data to validate the wood thrush (*Hylocichla mustelina*) model in their publication using Breeding Bird Survey (BBS) data. We were unable to validate the Baltimore oriole (*Icterus galbula*) and scarlet tanager (*Piranga olivacea*) model.

2.6. Illustrating applications

We applied the habitat model to each i-Tree plot, calculated an overall SI score (0–1) per species per i-Tree plot, calculated the mean SI score per species per city, and then calculated the mean SI score per land-use for each city. Although other land-uses were included in the i-Tree data collection, we focused on land-uses common for all ten cities: commercial, industrial, parks and forest, and residential. We also included vacant lots and transportation corridors, which were recorded in nine and eight of the ten cities, respectively. We describe the patterns of SI scores, land-uses, and management potential of i-Tree habitat models.

Although we did not directly test the effectiveness of habitat improvement plans, we demonstrated the potential of the i-Tree wildlife models to detect change in habitat conditions over time. For two cities (Baltimore, MD and Syracuse, NY), i-Tree data were collected at the same plot in 2001 and 2009. We used *t*-tests to determine whether the suitability for each land-use per city changed during the two data collection periods.

3. Results

3.1. Suitability index summaries

We developed 27 variable functions that were incorporated to form habitat models for nine species (Table 3). Overall, Moorestown, NJ had the highest quality habitat for birds (city-wide score for all species combined: 0.28), Jersey City, NJ the lowest (city-wide score: 0.14), and the remaining eight cities falling in between these SI scores (Table 4). On average, Philadelphia, PA had the highest SI score for Carolina chickadee, red-bellied woodpecker, and wood thrush while Jersey City had the lowest SI score for Baltimore oriole, Carolina chickadee, European starling, red-bellied woodpecker, scarlet tanager, and wood thrush (Table 4). Suitability within different land-uses varied for each species. Vacant lots, parks and forested land-uses had high SI scores for wood thrush, scarlet tanager, red-bellied woodpecker, and black-capped and Carolina chickadee. American robin had high SI scores for a variety of different land-uses and we did not discern any clear land-use signals. Industrial and commercial land-uses tended to score poorly with most species (Table 4).

3.2. Habitat model example: red-bellied woodpecker

The habitat suitability index model for the red-bellied woodpecker included four variables: tree density per 0.04 ha, basal area per ha, density of dead wood per 0.04 ha, and percent canopy cover per 0.04 ha. The species relies on forested areas and we included three variables to describe these habitat needs. Adkins Giese and Cuthbert (2003) observed 24 trees per 0.04 ha and a basal area of 34 m^2/ha in oak forests of the Upper Midwest, while Conner (1980) observed 30 trees/0.04 ha and a basal area of 14 m^2/ha in oak-hickory forests around Blacksburg, VA. However, these studies did not discern tree size. We wanted the model to reflect the mean diameter of the cavity limb (21.6 cm; Jackson, 1976) so only included trees greater than 23 cm dbh and adjusted the densities to reflect these conditions (Table 5). We fit a rational function $(0 - 0.0035 + (0.1606 \times \text{tree density}))/(1 + (-0.1417 \times \text{tree density}) + (0.0233 \times \text{tree density}^2))$ where tree density represents the density of trees greater than 23 cm dbh within a 0.04 ha plot, through these data points to predict how habitat suitability varied with large tree density (Fig. 1). We assumed suitability was the lowest when trees were absent. Our inclusion of basal area for all trees greater than 6 cm dbh reflects the propensity for this species to prefer relatively dense forests (Shackelford, Brown, & Conner, 2000; Table 6). We fit a logistic function $0.9906/(1 + (47.9216 \times \exp(-0.9689 \times \text{basal area})))$ where basal area is m^2/ha and calculated for all trees greater than 6 cm dbh, through these data points to quantify the relationship between basal area and the SI score (Fig. 2).

Canopy coverage has the potential to predict habitat suitability. DeGraaf, Yamasaki, Leak, and Lester (2006) suggested that when canopy coverage exceeds 35%, the site provided suitable conditions for red-bellied woodpeckers. We based our assumed values for canopy cover on qualitative accounts and personal observations of the species in forested suburban and riparian areas, with lack of observations in areas with little to no canopy cover and areas with an extremely dense canopy cover (Table 7). We fit a rational function $(-0.0371 + (0.0124 \times \text{percent canopy}))/(1 + (-0.0363 \times \text{percent canopy}) + (0.0005 \times \text{percent}$

Table 3
Habitat suitability equations for nine bird species in northeastern cities. Species codes as follows: AMRO, American robin; BAOR, Baltimore oriole; BCCH, black-capped chickadee; CACH, Carolina chickadee; EUST, European starling; NOCA, northern cardinal; RBWO, red-bellied woodpecker; SCTA, scarlet tanager; WOTH, wood thrush. Models with exp used base e.

Species	Variable (x)	Equation
AMRO	%TREE	$(0.6439054 + (-0.0023519694 \times x))/(1 + (-0.031238306 \times x) + (0.00059471346 \times x^2))$
AMRO	%GRASS M	$1/(4.19182 + (-0.083072 \times x) + (0.000538 \times x^2))$
BAOR	%TREE	$1.012735 \times \exp(0 - ((x - 35.4635207)^2)/(2 \times 15.3507889^2))$
BAOR	23cm_DENS	$(0.0377801 + (0.27942563 \times x))/(1 + (-0.4470676 \times x) + (0.13110269 \times x))$
BCCH	%TREE	$1.002 \times \exp((0 - ((x) - 63.568198)^2)/1795)$
BCCH	DEAD_DENS	$1.007/(1 + (32.567 \times \exp(-1.403x)))$
BCCH	MEAN_TOT HT_m	$0.97572/(1 + (11.742599 \times \exp(-0.48523169 \times)))$
CACH	%TREE	$1.002 \times \exp((0 - ((x) - 63.568198)^2)/1795)$
CACH	DEAD_DENS	$1.007/(1 + (32.567 \times \exp(-1.403x)))$
CACH	MEAN_TOT HT_m	$0.97572/(1 + (11.742599 \times \exp(-0.48523169 \times)))$
EUST	%BLDG	$(-0.00035052 + (0.0148132 \times x))/(1 + (-0.0378391 \times x) + (0.00065325 \times x^2)) \times -0.1$
EUST	DEAD_DENS	$0.800547 \times (1.2498289 - \exp(-2.42900485 \times x))$
EUST	%GRASS M	$1.02247/(1 + (40.643183849 \times \exp(-0.104376 \times x)))$
EUST	TR_DENS_ALL	$(0.81293 + (-0.0879822662 \times x))/(1 + (-0.3167288645 \times x) + (0.0546857954 \times x^2))$
NOCA	%TREE	$(0.63133686 + (-0.005359156 \times x))/(1 + (-0.036974589 \times x) + (0.0006728828 \times x^2))$
NOCA	%SHRB	$(0.00949075 + (0.021340335 \times x))/(1 + (-0.02120201 \times x) + (0.000432969 \times x^2))$
RBWO	BA_6 cm	$0.9906/(1 + (47.9216 \times \exp(-0.9689 \times x)))$
RBWO	%TREE	$(-0.0371 + (0.0124 \times x))/(1 + (-0.0335 \times x) + (0.0005 \times x^2)) \times -0.1$
RBWO	DEAD_DENS	$1/(1 + (15.67 \times \exp(-5.338 \times x)))$
RBWO	23cm_DENS	$(0 - 0.00347415 + (0.160609 \times x))/(1 + (-0.141679 \times x) + (0.0233308 \times x^2)) \times -0.1$
SCTA	BA_6 cm	$1.0363/(1 + (49.295 \times \exp(-0.1088 \times x)))$
SCTA	%TREE	$1.00545/(1 + (19{,}171.9801 \times \exp(-0.16936 \times x)))$
SCTA[a]	FOR_AREA	$((-0.0009840608 \times 4.3992415) + (1.6780139 \times x^{0.25391}))/(4.3992 + x^{0.2539122})$
SCTA	23cm_DENS	$1.01622702/(1 + (24{,}569.22035 \times \exp(-0.6493929 \times x)))$
WOTH[a]	FOR_1KM	$1.003/(1 + (224.7853 \times \exp(-0.1081 \times (x))))$
WOTH	%TREE	$1.03163/(1 + (141{,}241.64 \times \exp(-0.1531 \times x)))$
WOTH	SAP DENS	$(1.0401978/(1 + (65.800186 \times \exp(-0.758149 \times (x)))))$

[a] These models that used landscape variables were not included in the SI calculations but will be incorporated into the i-Tree program, and analyzed when spatial data is available.

Table 4
The suitability index (SI) scores for nine bird species in ten northeastern cities, for different urban land-uses. City SI score is the mean score per species and per city. Species codes as follows: AMRO, American robin; BAOR, Baltimore oriole; BCCH, black-capped chickadee; CACH, Carolina chickadee; EUST, European starling; NOCA, northern cardinal; RBWO, red-bellied woodpecker; SCTA, scarlet tanager; WOTH, wood thrush.

Land use	City	*n*	AMRO	BAOR	CACH	EUST	NOCA	RBWO	SCTA	WOTH	MEAN
CITY SI SCORE	Baltimore, MD	195	0.52	0.25	0.25	0.25	0.24	0.20	0.01	0.10	0.22
Commercial	Baltimore, MD	41	0.43	0.08	0.11	0.18	0.10	0.06	0.00	0.01	0.12
Industrial	Baltimore, MD	14	0.63	0.13	0.15	0.25	0.24	0.06	0.00	0.01	0.18
Park	Baltimore, MD	22	0.43	0.24	0.43	0.18	0.20	0.37	0.04	0.44	0.26
Residential	Baltimore, MD	90	0.57	0.33	0.26	0.35	0.32	0.22	0.01	0.06	0.27
Transportation	Baltimore, MD	16	0.45	0.23	0.32	0.09	0.16	0.29	0.03	0.15	0.20
Vacant	Baltimore, MD	5	0.51	0.49	0.26	0.07	0.55	0.26	0.00	0.02	0.24
CITY SI SCORE	Boston, MA	220	0.49	0.29	0.27	0.19	0.21	0.26	0.01	0.06	0.21
Commercial	Boston, MA	13	0.63	0.26	0.31	0.38	0.22	0.26	0.01	0.09	0.25
Industrial	Boston, MA	23	0.51	0.26	0.21	0.25	0.20	0.16	0.01	0.02	0.20
Park	Boston, MA	35	0.60	0.28	0.25	0.27	0.14	0.27	0.01	0.06	0.22
Residential	Boston, MA	62	0.47	0.41	0.36	0.13	0.30	0.41	0.01	0.09	0.26
Transportation	Boston, MA	10	0.51	0.11	0.13	0.12	0.11	0.07	0.00	0.00	0.12
Vacant	Boston, MA	28	0.34	0.24	0.50	0.01	0.21	0.49	0.03	0.22	0.23
CITY SI SCORE	Jersey City, NJ	230	0.47	0.11	0.15	0.18	0.16	0.04	0.00	0.01	0.14
Commercial	Jersey City, NJ	29	0.43	0.06	0.07	0.09	0.10	0.01	0.00	0.00	0.09
Industrial	Jersey City, NJ	4	0.39	0.05	0.06	0.01	0.08	0.01	0.00	0.00	0.07
Park	Jersey City, NJ	33	0.57	0.08	0.16	0.28	0.09	0.06	0.00	0.03	0.15
Residential	Jersey City, NJ	64	0.47	0.17	0.21	0.26	0.29	0.07	0.00	0.02	0.19
Transportation	Jersey City, NJ	25	0.46	0.08	0.16	0.06	0.10	0.02	0.00	0.00	0.10
Vacant	Jersey City, NJ	13	0.42	0.09	0.17	0.01	0.10	0.02	0.00	0.00	0.09
CITY SI SCORE	Moorestown, NJ	206	0.49	0.17	0.33	0.21	0.47	0.32	0.03	0.17	0.28
Commercial	Moorestown, NJ	31	0.50	0.18	0.20	0.20	0.66	0.14	0.01	0.03	0.25
Industrial	Moorestown, NJ	4	0.56	0.09	0.11	0.17	0.66	0.02	0.00	0.00	0.22
Park	Moorestown, NJ	45	0.44	0.07	0.41	0.18	0.35	0.41	0.08	0.33	0.28
Residential	Moorestown, NJ	103	0.56	0.25	0.34	0.28	0.50	0.33	0.02	0.10	0.31
Transportation	Moorestown, NJ	1	0.81	0.05	0.06	0.41	0.63	0.01	0.00	0.00	0.28
CITY SI SCORE	New York City	214	0.46	0.20	0.20	0.20	0.21	0.17	0.01	0.06	0.18
Commercial	New York City	6	0.84	0.20	0.13	0.42	0.22	0.05	0.00	0.00	0.21
Industrial	New York City	12	0.48	0.22	0.18	0.24	0.15	0.13	0.00	0.03	0.18
Park	New York City	33	0.45	0.13	0.26	0.17	0.19	0.29	0.02	0.13	0.19
Residential	New York City	76	0.50	0.35	0.25	0.32	0.28	0.27	0.01	0.03	0.26
Vacant	New York City	53	0.38	0.10	0.20	0.03	0.17	0.19	0.02	0.10	0.13
CITY SI SCORE	Philadelphia, PA	213	0.42	0.19	0.48	0.25	0.22	0.39	0.03	0.21	0.26
Commercial	Philadelphia, PA	3	0.75	0.41	0.30	0.54	0.20	0.29	0.00	0.00	0.29
Industrial	Philadelphia, PA	19	0.49	0.14	0.25	0.34	0.14	0.17	0.00	0.05	0.19
Park	Philadelphia, PA	53	0.28	0.13	0.74	0.10	0.17	0.69	0.07	0.54	0.30
Residential	Philadelphia, PA	62	0.57	0.33	0.40	0.52	0.26	0.30	0.00	0.02	0.31
Transportation	Philadelphia, PA	10	0.44	0.17	0.20	0.08	0.24	0.09	0.00	0.00	0.14
Vacant	Philadelphia, PA	50	0.31	0.10	0.54	0.03	0.26	0.42	0.03	0.29	0.22
CITY SI SCORE	Scranton, PA	191	0.50	0.20	0.25	0.23	0.23	0.22	0.01	0.16	0.22
Commercial	Scranton, PA	32	0.47	0.15	0.10	0.16	0.20	0.05	0.00	0.01	0.14
Industrial	Scranton, PA	11	0.49	0.15	0.10	0.19	0.15	0.04	0.00	0.00	0.14
Park	Scranton, PA	9	0.54	0.29	0.33	0.25	0.25	0.35	0.01	0.29	0.26
Residential	Scranton, PA	94	0.56	0.18	0.19	0.44	0.22	0.13	0.01	0.06	0.23
Transportation	Scranton, PA	13	0.44	0.16	0.16	0.05	0.18	0.10	0.00	0.03	0.13
Vacant	Scranton, PA	29	0.26	0.10	0.53	0.02	0.17	0.48	0.03	0.61	0.25
CITY SI SCORE	Syracuse, NY	200	0.58	0.18	0.29	0.30	0.25	0.14	0.00	0.12	0.23
Commercial	Syracuse, NY	15	0.45	0.11	0.14	0.22	0.18	0.07	0.00	0.00	0.15
Industrial	Syracuse, NY	18	0.57	0.11	0.27	0.27	0.16	0.08	0.00	0.22	0.21
Park	Syracuse, NY	7	0.67	0.26	0.17	0.42	0.12	0.19	0.00	0.01	0.21
Residential	Syracuse, NY	113	0.64	0.20	0.24	0.38	0.26	0.12	0.00	0.03	0.24
Transportation	Syracuse, NY	9	0.50	0.13	0.22	0.07	0.51	0.04	0.00	0.01	0.17
Vacant	Syracuse, NY	30	0.40	0.11	0.50	0.10	0.17	0.21	0.01	0.46	0.23
CITY SI SCORE	Washington, DC	201	0.50	0.31	0.26	0.23	0.22	0.31	0.07	0.06	0.23
Commercial	Washington, DC	10	0.43	0.15	0.12	0.17	0.21	0.09	0.00	0.00	0.15
Industrial	Washington, DC	7	0.46	0.19	0.10	0.16	0.21	0.08	0.00	0.00	0.15
Park	Washington, DC	53	0.46	0.24	0.33	0.15	0.24	0.41	0.17	0.15	0.24
Residential	Washington, DC	91	0.50	0.44	0.27	0.20	0.30	0.36	0.03	0.03	0.26
CITY SI SCORE	Woodbridge, NJ	215	0.52	0.23	0.27	0.21	0.07	0.24	0.01	0.12	0.20
Commercial	Woodbridge, NJ	20	0.45	0.19	0.16	0.14	0.08	0.14	0.01	0.06	0.15
Industrial	Woodbridge, NJ	5	0.43	0.09	0.09	0.01	0.09	0.01	0.00	0.00	0.08
Park	Woodbridge, NJ	29	0.32	0.10	0.56	0.13	0.03	0.59	0.07	0.48	0.25
Residential	Woodbridge, NJ	98	0.64	0.35	0.27	0.32	0.08	0.24	0.01	0.04	0.24
Transportation	Woodbridge, NJ	22	0.50	0.11	0.13	0.13	0.08	0.04	0.00	0.05	0.12

Table 5
Relationship between large tree density (trees larger than 23 cm dbh) per 0.04 ha and suitability index (SI) for red-bellied woodpecker (RBWO) habitat, and associated references.

Large tree density (per 0.04 ha)	SI score (RBWO)	Reference
0	0	Assumed value
3	0.6	Assumed value
6	1	Adkins Giese and Cuthbert (2003)
8	0.9	Conner (1980)
11	0.8	Assumed value

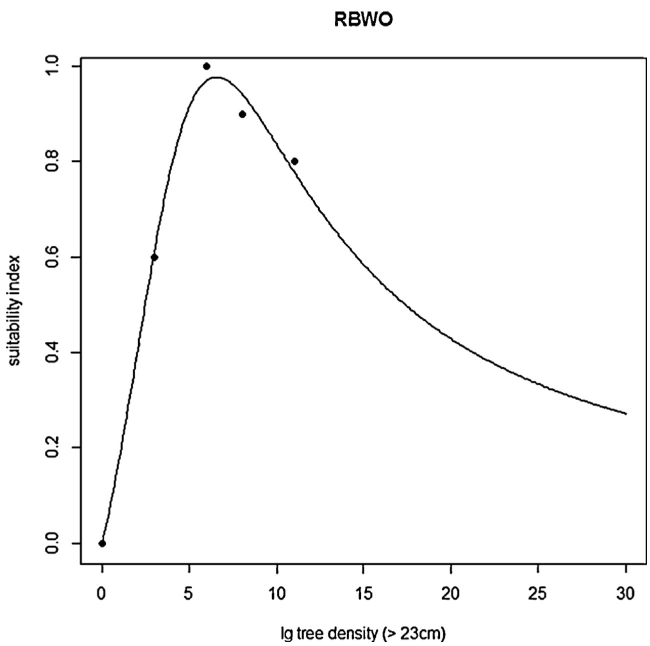

Fig. 1. Relationship between large tree density (trees larger than 23 cm dbh) per 0.04 ha and suitability index (SI) for red-bellied woodpecker (RBWO) habitat, and associated references.

Table 6
Relationship between basal area (trees > 6 cm dbh) per ha and suitability index (SI) for red-bellied woodpecker (RBWO) habitat, and associated references.

Basal area (per ha)	SI score (RBWO)	Reference
0	0	Assumed value
4	0.5	Assumed value
8	0.95	Conner, 1980 (based on SD)
14	1	Conner, 1980
34	1	Adkins Giese and Cuthbert (2003)

canopy^2)), where percent canopy represents the percent of a 0.04 ha plot with tree canopy cover, through these data points to predict how habitat suitability varied with canopy coverage (Fig. 3). We assumed suitability was the lowest when trees were absent.

Table 7
Relationship between canopy percent per 0.04 ha and suitability index (SI) for red-bellied woodpecker (RBWO) habitat, and associated references.

Canopy percent (per 0.04 ha)	SI score (RBWO)	Reference
0	0	Assumed value
15	0.1	Assumed value
20	0.3	Assumed value
25	0.5	Assumed value
35	0.9	DeGraaf et al. (2006)
62	1	Straus et al. (2011)

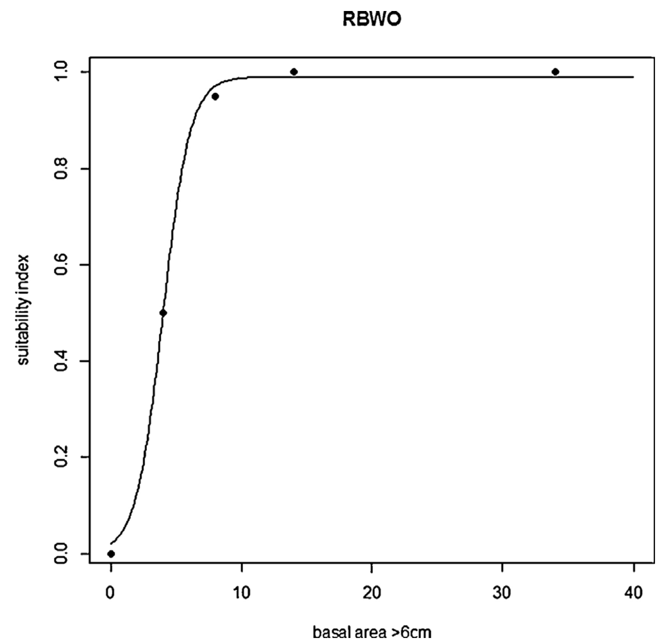

Fig. 2. Relationship between basal area (trees > 6 cm dbh) per ha and suitability index (SI) for red-bellied woodpecker (RBWO) habitat, and associated references.

Although dead wood is necessary for foraging and nesting, it is not essential for detecting red-bellied woodpeckers. Of 42 nests in southwest Ontario, Straus, Bavrlic, Nol, Burke, and Elliott (2011) observed 93% of the nests in dead and declining trees and 6% of nests in healthy trees. Adkins Giese and Cuthbert (2003) observed three dead or declining trees per 0.04 ha in the Midwest (Table 8). We fit a logistic function $1/(1+(15.67 \times \exp(-5.338 \times \text{dead wood density per } 0.04\text{ ha})))$ (where dead wood is recorded as trees with a condition of fair, poor, dying or dead) through these data points to quantify the relationship between trees with dead wood and the SI score (Fig. 4). We calculated the geometric mean of these habitat models to generate a final SI score for this species.

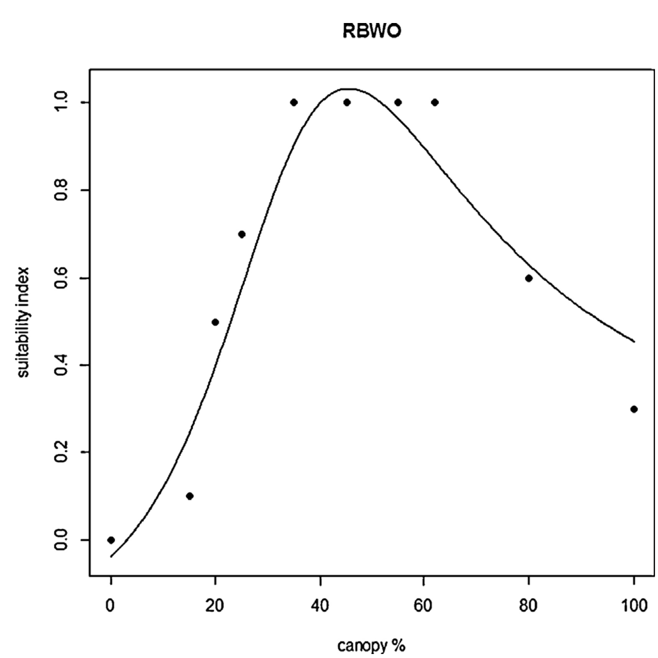

Fig. 3. Relationship between canopy percent per 0.004 ha and suitability index (SI) for red-bellied woodpecker (RBWO) habitat, and associated references.

Table 8
Relationship between dead wood density per ha and suitability index (SI) for red-bellied woodpecker (RBWO) habitat, and associated references.

Dead wood density (per 0.04 ha)	SI score (RBWO)	Reference
0	0.06	Straus et al. (2011)
1	0.93	Straus et al. (2011)
3	1	Adkins Giese and Cuthbert (2003)

3.3. Model validations

At the BES LTER sites, the American robin was recorded in 72 of the 83 bird monitoring/i-Tree locations, Carolina chickadee in 19 of the 83 locations, European starling in 62 of the 83 locations, northern cardinal in 60 of the 83 locations, and red-bellied woodpecker in 12 of the 83 locations. Spearman rank correlation identified a significant and positive relationship between the HSI score and mean bird abundance at the BES LTER i-Tree locations for American robin ($P=0.0043$, $r_s=0.31$), Carolina chickadee ($P=0.0011$, $r_s=0.3515$), northern cardinal ($P=0.0022$, $r_s=0.3311$), red-bellied woodpecker ($P=0.0008$, $r_s=0.3596$), and European starling ($P=0.0349$, $r_s=0.2333$). When testing the sensitivity of the models by subsequently removing individual variables from whole models, we found no discrepancies between these partial and full models in their ability to predict mean bird abundance better than chance for Carolina chickadee, European starling and red-bellied woodpecker. The spearman rank correlation did not detect a significant relationship between the HSI score and mean bird abundance in the American robin model when lawn percent was omitted ($P=0.5976$, $r_s=0.0593$). However, when the model omitted canopy cover and included lawn percent, we found a significant relationship between the HSI score and mean abundance ($P=0.0071$, $r_s=0.2950$). Similarly, when the percent shrub cover was removed from the northern cardinal model, the model failed to predict presence when this species was recorded, though a model with just percent shrubs was significant ($P=0.0140$, $r_s=0.2705$).

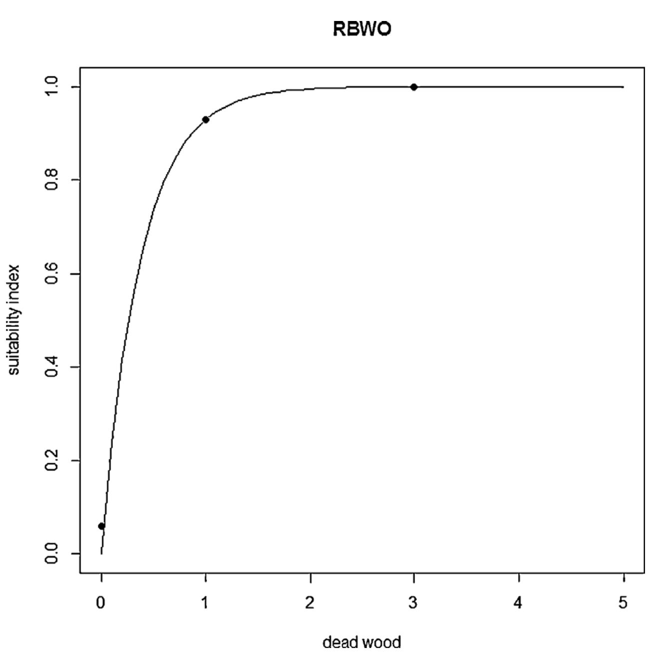

Fig. 4. Relationship between deadwood density per ha and suitability index (SI) for red-bellied woodpecker (RBWO) habitat, and associated references.

3.4. Illustrating applications

For the most part, habitat suitability in Baltimore and Syracuse declined from 2001 to 2009 (Table 9). Important resources such as canopy cover in Baltimore declined by 33.8% in vacant lots, and large tree density in Syracuse declined by 0.8 and 3.4 trees in residential and vacant lots between the two time periods (*unpublished i-Tree dataset*). Habitat suitability scores significantly decreased for Baltimore oriole, northern cardinal, and red-bellied woodpecker between 2001 and 2009 in Syracuse residential areas and vacant lots, and for scarlet tanagers in vacant lots only. Habitat suitability also differed for red-bellied woodpecker in Baltimore residential areas and for Carolina chickadee, red-bellied woodpecker, and wood thrush in Baltimore vacant lots. In contrast, habitat suitability increased for wood thrushes (Syracuse) and northern cardinals (Baltimore) in residential areas during this time period (Table 9). We failed to find a significant change in commercial, cemetery, golf course or institutional land-use plots in Baltimore and Syracuse.

4. Discussion

Integrating validated bird habitat suitability models into i-Tree can provide a more comprehensive assessment of the ecosystem services provided by the urban forest. Essentially, our models translate the i-Tree raw data's detailed information on the forest composition and structure into relative assessments of habitat value for birds. The bird habitat models suggest which species specifically, and guilds broadly, can be supported by an urban forest. By selecting which bird models to focus on (e.g., native or rare species), other societal values can be included in this assessment and guide general forest planning in urban areas. In addition, the bird habitat models have the capacity to provide specific targets (i.e., canopy percent or dead wood density) geared toward urban foresters and planners when determining how to manage the urban forest for wildlife.

Our validation efforts support the efficacy of using the habitat models to predict the habitat quality of urban areas for a variety of species. Although we were unable to validate the Baltimore oriole and scarlet tanager model at this time, we agree with Brooks (1997) that these untested models still have greater value than no information about these species' habitat relationships. In several cases, sensitivity analyses helped to identify particularly influential habitat parameters. For example, percent lawn for American robin and percent shrub cover for northern cardinal have strong influences on the habitat suitability for the respective species. Although the models with insignificant results highlight the unequal effect of these particular variables, the models that included all the habitat variables had a higher rank scores, suggesting the model had stronger predictive power when these variables were included.

The i-Tree habitat models link habitat features with an SI score reflecting the suitability of a site for that species. Each habitat variable has an optimal value for a particular species (i.e., when the suitability index score is 1.0, the site has the greatest potential to support said species). Less than optimal values result in lower SI scores and provide a baseline for habitat improvement recommendations. Compared with the other cities, Jersey City had the lowest mean SI scores for all but one species (Table 3). The i-Tree program assessed canopy coverage at 13%, well below the national average of 35.1% (Nowak & Greenfield, 2012). Eight species included canopy percent as an important limiting variable with optimal values ranging between 25% and 100% (Supplementary material).

Urban parks, vacant lots, and residential land-uses had high SI scores for most of the species modeled (Table 3), and species of conservation concern in particular (Dettmers & Rosenberg, 2000). For example, urban parks and vacant lots had the highest SI score for

Table 9
A comparison of suitability index (SI) scores for six bird species and mean values for two habitat variables at the same i-Tree monitoring plot in 2001 and 2009 in Syracuse, NY and Baltimore, MD for residential and vacant lot land-uses. The SI scores for American robin and European starling did not exhibit any significant changes. Species habitat models in commercial and institutional land-uses, and golf courses failed to show significant relationships.

	Residential				Vacant lot			
	2001	2009	*F*	*P*	2001	2009	*F*	*P*
BALTIMORE	*n* = 87	*n* = 90			*n* = 18	*n* = 5		
American robin	0.54	0.57	1.22	0.27	0.39	0.51	1.62	0.22
Baltimore oriole	0.35	0.33	0.09	0.75	0.35	0.49	0.58	0.45
Carolina chickadee	0.3	0.26	1.34	0.25	0.65	0.26	5.28	0.032
European starling	0.34	0.3	0.88	0.35	0.07	0.04	0.24	0.63
Northern cardinal[a]	0.35	0.33	0.47	0.49	0.22	0.55	16.71	0.0005
Red-bellied woodpecker	0.34	0.24	4.28	0.04	0.66	0.26	5.53	0.029
Scarlet tanager	0.01	0.01	0.97	0.33	0.1	0.01	2.05	0.17
Wood thrush	0.03	0.06	2.19	0.14	0.3	0.14	3.30	0.084
Tree canopy	25.31	24.74	0.02	0.88	52.22	18.4	5.50	0.03
SYRACUSE	*n* = 117	*n* = 113			*n* = 33	*n* = 30		
American robin	0.61	0.64	1.24	0.27	0.37	0.4	0.27	0.6
Baltimore oriole	0.46	0.22	46.37	<0.001	0.22	0.11	3.62	0.06
Black-capped chickadee	0.23	0.27	2.64	0.11	0.48	0.5	0.03	0.86
European starling	0.28	0.32	2.38	0.12	0.05	0.02	1.05	0.31
Northern cardinal	0.39	0.29	11.92	0.0007	0.32	0.17	6.81	0.01
Red-bellied woodpecker	0.24	0.16	7.57	0.0064	0.5	0.21	14.53	0.0003
Scarlet tanager	0.01	0	0.42	0.52	0.07	0.01	5.24	0.026
Wood thrush[a]	0.01	0.04	6.01	0.015	0.36	0.46	0.83	0.37
Large tree density	1.16	0.39	24.88	<0.0001	3.85	0.4	22.46	<0.0001

[a] An increase in suitability.

scarlet tanager and wood thrush, suggesting that when managed for wildlife, these urban land-uses have the potential to support rare species. Residential land-uses had the highest SI score for Baltimore oriole (Table 3) and although this land-use scored low for wood thrush, the patterns suggest the existence of potential habitat and the conservation value of residential areas (Lerman & Warren, 2011).

The active management of dead wood in urban areas has the potential to stabilize populations for a guild that often adapts well to cities (Chace & Walsh, 2006). Urban parks in Boston, MA and New York City had low SI scores compared to urban parks in Philadelphia, PA for red-bellied woodpecker, an obligate cavity nester. Boston and New York also had low densities of dead wood, an important nesting resource for the species (Shackelford et al., 2000). On average, Boston had 0.66 trees with dead wood (Dead Dens) per plot (6% of trees had some dead wood; *unpublished i-Tree dataset*) and New York City had 0.85 trees with dead wood per plot (6% of trees had some dead wood; Nowak, Hoehn, Crane, Stevens, & Walton, 2007). The model for dead wood density calculated an SI score of 1 (i.e., most suitable) when at least three trees with dead wood were present in a 0.04 ha plot. The model calculated an SI score of 0.93 with at least one tree with dead wood. Based on the dead wood present, these two cities failed to reach a suitability threshold that had a high likelihood of supporting species requiring dead wood (i.e., areas with at least one tree with dead wood) whereas Philadelphia, with an average nine trees per plot with dead wood (57% of all trees; *unpublished i-Tree dataset*), had a greater potential to support this species because of the presence of an important resource for cavity nesting species. Black-capped chickadee, an additional species belonging to this nesting guild, had similar patterns.

The differences in dead wood densities might be the result of different management regimes for these cities. Perhaps the former two cities have a more active urban forestry department and remove a greater degree of dead wood due to the hazards and esthetics associated with dead and dying limbs (Harris, Clark, & Matheny, 2004). Alternatively, the differences could also be due to different tree population structures (e.g., age or size distribution) among cities. By delineating a threshold of suitability for each habitat variable, the models provide specific targets for improving the habitat conditions for a particular species, which is necessary for identifying management goals (Kroll & Haufler, 2006). For example, the city of New York had low scores for red-bellied woodpecker, particularly in commercial and industrial land-uses. Based on the habitat model description for this species (see model example), the optimal values for key habitat features are as follows: six large trees (> 23 cm dbh) per 0.04 ha, 14 m^2/ha basal area, 35–62% canopy coverage per 0.04 ha, and at least three trees with dead wood within 0.04 ha (Tables 1–4, respectively). Managers can then review the i-Tree data and assess how well the actual habitat values accord with the optimal values. In New York City forest patches, the canopy percentage reached optimal values though the amount of deadwood fell below the threshold (*unpublished i-Tree dataset*). Thus incorporating management initiatives that encourage dead wood would improve the habitat conditions for this and other cavity nesting species. In sum, when cities or land-uses have low SI scores, the manager can pinpoint the sub-optimal variables and develop management plans that target these low scoring habitat features.

Our example of how the i-Tree habitat module can document SI changes over time demonstrated the potential for assessing the effectiveness of management plans (or lack thereof). For example, in the Baltimore i-Tree dataset, we noted a sharp decline of trees with dead wood between 2001 (3.59 trees per i-Tree plot) and 2009 (0.73 trees per i-Tree plot). The deadwood density threshold for a suitable site for red-bellied woodpecker was three. Therefore this loss of deadwood might explain why the suitability index for species that rely on this resource also declined. An effective management strategy would include more selective criteria for removing dead wood (e.g., only when posing a strong hazard risk), or perhaps encouraging the development and retention of snags in areas not frequented by people.

The models provide a substantial initial assessment of the habitat potential in the urban forest, while assisting decision makers with the ultimate goal of improving urban bird habitat (Beaudry et al., 2010). Although the number of studies focusing on urban birds has increased over the past 20 years (Ramalho & Hobbs, 2012), and many of these studies included recommendations on how to improve urban habitat, the recommendations are often for a specific city (Lerman & Warren, 2011), and not necessarily accessible to managers. The i-Tree tool was designed for urban managers and thus the wildlife component expands the capacity of the tool to allow for a more comprehensive assessment of the ecosystem services provided by the urban forest. With rapid habitat suitability assessment capabilities and ease of use for non-professional scientists, the wildlife component of i-Tree delivers a valuable tool that is applicable on a regional scale.

We recognize the importance of local and landscape features in limiting urban bird distribution (Chamberlain, Cannon, & Toms, 2004; McCaffrey & Mannan, 2012). We did not have spatial locations available for the majority of the i-Tree plots and thus did not incorporate these landscape variables into the SI calculations. However, landscape variables are known to influence the distribution for two of our modeled species: scarlet tanager and wood thrush (Hoover & Brittingham, 1998; Robinson, Thompson, Donovan, Whitehead, & Faaborg, 1995). We describe these models based on landscape features (e.g., percent forest cover within 1 km radius of i-Tree plot; Table 3), and will include the models in the i-Tree program when spatial data are available.

Although currently limited to the local scale, the i-Tree habitat models have the advantage of calculating SI for specific land-uses, a known feature that influences urban bird distribution (Blair, 1996), and thus enabling managers to target low-scoring land-uses independently. By discriminating among the land-use differences, the tool recognizes the different jurisdictions and land ownership, and the associated management strategies. For example, the strategy for increasing canopy coverage in city-owned open space might differ from residential lands, since the latter might require participation from private households and the former might require public support for urban forestry programs (Warren, Ryan, Lerman, & Tooke, 2011). This local scale also provides greater opportunities for intervention. For example, mangers can affect canopy percentage through tree planting efforts but have little opportunity to significantly increase the area of forest tracts embedded within the urban matrix. Thus, although protecting large tracts of contiguous forest is essential for forest interior species (Robinson et al., 1995), once the land becomes developed, there is little chance to effectively manage and incorporate management improvement plans at this scale.

Similar to other habitat models, the i-Tree habitat models were not as robust for generalist species compared with habitat specialists (Tirpak, Jones-Farrand, Thompson, Twedt, Baxter, et al., 2009). For example, the European starling, an urban exploiter (Blair, 1996), scored lower than expected for each city in all the urban land-uses (Table 4), indicating that the ten cities used in the habitat model demonstration supported few starlings. Based on personal observations and the numerous studies documenting starlings as one of the most abundant urban birds (Chace & Walsh, 2006), we can assume that the model did not accurately reflect starling habitat suitability. This was further supported during the validation process. The results from our models also suggested that variables other than those measured using i-Tree might better explain the habitat suitability of this ubiquitous species. Habitat specialists by their very nature are more restricted to a few key habitat features (Kilgo et al., 2002). The i-Tree habitat models also had the tendency to overestimate the suitability of potential habitat. The model calculated a high likelihood of occupancy (>0.5) for more sites than will be occupied since the models did not account for interspecific competition, an additional factor that limits distribution (Fielding & Bell, 1997; Shochat et al., 2010).

Future directions include integrating these models into the i-Tree program which involves coding the equations in i-Tree Eco. We plan to generate GIS range maps for each species to identify the regions these equations should be activated (based on Breeding Bird Survey data). We plan to model additional species in other regions, identify additional variables for the i-Tree data collection protocol that will help improve the estimation of the SI, and collect bird abundance data at i-Tree plots to further validate the models. We also urge future urban bird studies to adapt a habitat assessment protocol that includes the i-Tree variables and data collection at the same spatial scale (0.04 ha). These studies will enable us to further model validation efforts as well as compare urban bird habitats among cities.

The i-Tree habitat models provide a tool for local or regional initial assessments of the current state of the urban forest for providing bird habitat. The assessment can be the basis for an extensive and comprehensive conservation plan specifically geared toward urban land-uses. Results from this study will help guide urban foresters, planners, and landscape designers who require specific information such as how many trees and shrubs are necessary within an urban greening project to reach conservation goals targeted at improving the suitability of urban bird habitat. Given that more than 80% of Americans live in urban environments (US Census, 2012), it becomes imperative that urban forests provide opportunities for urban dwellers to connect with nature. This connection can improve and enhance health and well-being (Fuller, Irvine, Devine-Wright, Warren, & Gaston, 2007) while generating interest and support for conservation initiatives that aim to improve urban biodiversity (Miller, 2005).

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.landurbplan.2013.10.006.

References

Adams, C. C. (1935). The relation of general ecology to human ecology. *Ecology*, *16*(3), 316–335.

Adkins Giese, C. L., & Cuthbert, F. J. (2003). Influence of surrounding vegetation on woodpecker nest tree selection in oak forests of the Upper Midwest, USA. *Forest Ecology and Management 179*, 523–534.

Akbari, H., Pomerantz, M., & Taha, H. (2001). Cool surfaces and shade trees to reduce energy use and improve air quality in urban areas. *Solar Energy*, *70*(3), 295–310.

Beaudry, F., Pidgeon, A. M., Radeloff, V. C., Howe, R. W., Mladenoff, D. J., & Bartelt, G. A. (2010). Modeling regional-scale habitat of forest birds when land management guidelines are needed but information is limited. *Biological Conservation*, *143*(7), 1759–1769. http://dx.doi.org/10.1016/j.biocon.2010.04.025

Beissinger, S. R., & Osborne, D. R. (1982). Effects of urbanization on avian community organization. *Condor*, *84*(1), 75–83.

Berlanga, H., Kennedy, J. A., Rich, T. D., Arizmendi, M. C., Beardmore, C. J., Blancher, P. J., et al. (2010). *Saving our shared birds: Partners in flight tri-national vision for landbird conservation*. Ithaca, NY: Cornell Lab of Ornithology. Retrieved from:. http://www.savingoursharedbirds.org/final_reports_pdfs/PIF2010_English_Final.pdf

Blair, R. B. (1996). Land use and avian species diversity along an urban gradient. *Ecological Applications*, *6*(2), 506–519.

Brooks, R. P. (1997). Improving habitat suitability index models. *Wildlife Society Bulletin*, *25*(1), 163–167. http://dx.doi.org/10.2307/3783299
Caro, T., & O'Doherty, G. (1999). On the use of surrogate species in conservation biology. *Conservation Biology*, *13*(4), 805–814. http://dx.doi.org/10.1046/j.1523-1739.1999.98338.x
Chace, J. F., & Walsh, J. J. (2006). Urban effects on native avifauna: A review. *Landscape and Urban Planning*, *74*(1), 46–69.
Chamberlain, D. E., Cannon, A. R., & Toms, M. P. (2004). Associations of garden birds with gradients in garden habitat and local habitat. *Ecography*, *27*(5), 589–600.
Clergeau, P., Savard, J. P. L., Mennechez, G., & Falardeau, G. (1998). Bird abundance and diversity along an urban-rural gradient: A comparative study between two cities on different continents. *Condor*, *100*(3), 413–425.
Conner, R. (1980). Foraging habits of woodpeckers in southwestern Virginia. *Journal of Field Ornithology*, *51*(2), 119–127.
Croci, S., Butet, A., & Clergeau, P. (2008). Does urbanization filter birds on the basis of their biological traits? *Condor*, *110*(2), 223–240.
Crooks, K. R., Suarez, A. V., & Bolger, D. T. (2004). Avian assemblages along a gradient of urbanization in a highly fragmented landscape. *Biological Conservation*, *115*(3), 451–462.
Czech, B., Krausman, P. R., & Devers, P. K. (2000). Economic associations among causes of species endangerment in the United States. *BioScience*, *50*(7), 593–601.
DeGraaf, R., Tilghman, N., & Anderson, S. (1985). Foraging guilds of North-American birds. *Environmental Management*, *9*(6), 493–536.
DeGraaf, R. M., & Wentworth, J. M. (1986). Avian guild structure and habitat associations in suburban bird communities. *Urban Ecology*, *9*(3/4), 399–412.
DeGraaf, R. M., Yamasaki, M., Leak, W. B., & Lester, A. M. (2006). *Technical guide to forest wildlife habitat management in New England*. Burlington, Vermont: University of Vermont Press.
Dettmers, R., & Rosenberg, K. V. (2000). *Partners in Flight landbird conservation plan: physiographic area 9: Southern New England*. Retrieved from:. http://www.partnersinflight.org/bcps/plan/pl_09_10.pdf
Driscoll, C. T., Lambert, K. F., Chapin, F. S., III, Nowak, D. J., Spies, T. A., Swanson, F. J., et al. (2012). Science and society: The role of long-term studies in environmental stewardship. *BioScience*, *62*(4), 354–366.
eBird. (2012). *eBird: An online database of bird distribution and abundance [web application]*. Ithaca, NY: eBird. Retrieved from:. http://www.ebird.org
Emlen, J. T. (1974). Urban bird community in Tucson, Arizona – Derivation, structure, regulation. *Condor*, *76*(2), 184–197.
Fernandez-Juricic, E. (2000). Avifaunal use of wooded streets in an urban landscape. *Conservation Biology*, *14*(2), 513–521. http://dx.doi.org/10.1046/j.1523-1739.2000.98600.x
Fielding, A. H., & Bell, J. F. (1997). A review of methods for the assessment of prediction errors in conservation presence/absence models. *Environmental Conservation*, *24*(1), 38–49.
Fitzgerald, J., Thogmartin, W. E., Dettmers, R., Jones, T., Rustay, C., Ruth, J. M., et al. (2009). Application of models to conservation planning for terrestrial birds in North America. In J. J. Millspaugh, & F. R. Thompson III (Eds.), *Models for planning wildlife conservation in large landscapes* (pp. 593–624). Boston, MA, USA: Academic Press.
Fuller, R. A., Irvine, K. N., Devine-Wright, P., Warren, P. H., & Gaston, K. J. (2007). Psychological benefits of greenspace increase with biodiversity. *Biology Letters*, *3*(4), 390–394.
Gering, J., & Blair, R. (1999). Predation on artificial bird nests along an urban gradient: Predatory risk or relaxation in urban environments? *Ecography*, *22*(5), 532–541.
Goddard, M., Dougill, A., & Benton, T. (2010). Scaling up from gardens: Biodiversity conservation in urban environments. *Trends in Ecology & Evolution*, *25*(2), 90–98. http://dx.doi.org/10.1016/j.tree.2009.07.016
Grinnell, J. (1917). The niche-relationships of the California Thrasher. *Auk*, *34*(4), 427–433.
Harris, R. W., Clark, J. R., & Matheny, N. P. (2004). *Arboriculture: Integrated management of landscape trees, shrubs, and vines*. Upper Saddle River: Prentice Hall.
Hoover, J. P., & Brittingham, M. C. (1998). Nest-site selection and nesting success of wood thrushes. *Wilson Bulletin*, *110*(3), 375–383.
Jackson, J. A. (1976). A comparison of some aspects of the breeding ecology of red-headed and red-bellied woodpeckers in Kansas. *Condor*, *78*(1), 67–76. http://dx.doi.org/10.2307/1366917
Jones-Farrand, D. T., Fearer, T. M., Thogmartin, W. E., Thompson, F. R. T., III, Nelson, M. D., & Tirpak, J. M. (2011). Comparison of statistical and theoretical habitat models for conservation planning: The benefit of ensemble prediction. *Ecological Applications*, *21*(6), 2269–2282. http://dx.doi.org/10.1890/10-1047.1
Kilgo, J. C., Gartner, D. L., Chapman, B. R., Dunning, J. B., Jr., Franzreb, K. E., Gauthreaux, S. A., et al. (2002). A test of an expert-based bird-habitat relationship model in South Carolina. *Wildlife Society Bulletin*, *30*(3), 783–793.
Kinzig, A. P., Warren, P., Martin, C., Hope, D., & Katti, M. (2005). The effects of human socioeconomic status and cultural characteristics on urban patterns of biodiversity. *Ecology and Society*, *10*(1).
Kroll, A. J., & Haufler, J. B. (2006). Development and evaluation of habitat models at multiple spatial scales: A case study with the dusky flycatcher. *Forest Ecology and Management*, *229*(1–3), 161–169. http://dx.doi.org/10.1016/j.foreco.2006.03.026
Lack, D. (1933). Habitat selection in birds. With special reference to the effects of afforestation on the Breckland avifauna. *Journal of Animal Ecology*, *2*(2), 239–262.
Lerman, S. B., Turner, V. K., & Bang, C. (2012). Homeowner associations as a vehicle for promoting native urban biodiversity. *Ecology and Society*, *17*(4) http://dx.doi.org/10.5751/ES-05175-170445
Lerman, S. B., & Warren, P. S. (2011). The conservation value of residential yards: Linking birds and people. *Ecological Applications*, *21*(4), 1327–1339.
McCaffrey, R. E., & Mannan, R. W. (2012). How scale influences birds' responses to habitat features in urban residential areas. *Landscape and Urban Planning*, *105*(3), 274–280. http://dx.doi.org/10.1016/j.landurbplan.2011.12.022
McCune, B., & Grace, J. B. (2002). *Analysis of ecological communities*. Gleneden Beach, OR: MjM Software Design.
Melles, S. J. (2005). Urban birds diversity as an indicator of human social diversity and economic inequality in Vancouver, British Columbia. *Urban Habitats*, *3*(1), 25–48.
Miller, J. R. (2005). Biodiversity conservation and the extinction of experience. *Trends in Ecology & Evolution*, *20*(8), 430–434.
Miller, J. R., & Hobbs, R. J. (2002). Conservation where people live and work. *Conservation Biology*, *16*(2), 330–337. http://dx.doi.org/10.1046/j.1523-1739.2002.00420.x
Nilon, C. H., Warren, P. S., & Wolf, J. (2011). Baltimore birdscape study: Identifying habitat and land-cover variables for an urban bird-monitoring project. *Urban Habitats*, *6*. Retrieved from:. http://www.urbanhabitats.org/v06n01/baltimore_full.html
Nowak, D. J., Crane, D. E., & Stevens, J. C. (2006). Air pollution removal by urban trees and shrubs in the United States. *Urban Forestry & Urban Greening*, *4*(3), 115–123.
Nowak, D. J., Crane, D. E., Stevens, J. C., Hoehn, R. E., Walton, J. T., & Bond, J. (2008). A ground-based method of assessing urban forest structure and ecosystem services. *Arboriculture and Urban Forestry*, *34*(6), 347–358.
Nowak, D. J., & Greenfield, E. J. (2012). Tree and impervious cover in the United States. *Landscape and Urban Planning*, *107*(1), 21–30.
Nowak, D. J., Greenfield, E. J., Hoehn, R. E., & Lapoint, E. (2013). Carbon storage and sequestration by trees in urban and community areas of the United States. *Environmental Pollution*, *178*, 229–236.
Nowak, D. J., Hirabayshi, S., Bodine, A., & Hoehn, R. (2013). Modeled PM2. 5 removal by trees in ten U.S. cities and associated health effects. *Environmental Pollution*, *178*, 395–402.
Nowak, D. J., Hoehn, R., Crane, D. E., Stevens, J. C., & Walton, J. T. (2007). *Assessing urban forest effects and values: New York City's urban forest. USDA Forest Service, Northern Research Station Resource Bulletin NRS-9*. Newtown Square: PA.
Nowak, D. J., & Walton, J. T. (2005). Projected urban growth (2000–2050) and its estimated impact on the US forest resource. *Journal of Forestry*, *103*(8), 383–389.
Ramalho, C. E., & Hobbs, R. J. (2012). Time for a change: Dynamic urban ecology. *Trends in Ecology & Evolution*, *27*(3), 179–188. http://dx.doi.org/10.1016/j.tree.2011.10.008
Rich, T. D., Beardmore, C. J., Berlanga, H., Blancher, P. J., Bradstreet, M. S. W., & Butcher, G. S. (2004). *Partners in Flight North American landbird conservation plan*. Ithaca, NY: Cornell Lab of Ornithology. Retrieved from:. http://www.partnersinflight.org/cont_plan/
Robinson, S. K., Thompson, F. R., III, Donovan, T. M., Whitehead, D. R., & Faaborg, J. (1995). Regional forest fragmentation and the nesting success of migratory birds. *Science*, *267*(5206), 1987.
Roloff, G. J., & Kernohan, B. J. (1999). Evaluating reliability of habitat suitability index models. *Wildlife Society Bulletin*, *27*(4), 973–985.
Shackelford, C. E., Brown, R. E., & Conner, R. N. (2000). Red-bellied woodpecker (*Melanerpes carolinus*). In A. Poole (Ed.), *The Birds of North America Online* http://dx.doi.org/10.2173/bna.500. Retrieved from:. http://bna.birds.cornell.edu/bna/species/500
Shanahan, D. F., Possingham, H. P., & Martin, T. G. (2011). Foraging height and landscape context predict the relative abundance of bird species in urban vegetation patches. *Austral Ecology*, *36*(8), 944–953. http://dx.doi.org/10.1111/j.1442-9993.2010.02225.x
Shochat, E., Lerman, S. B., Anderies, J. M., Warren, P. S., Faeth, S. H., & Nilon, C. H. (2010). Invasion, competition, and biodiversity loss in urban ecosystems. *BioScience*, *60*(3), 199–208. http://dx.doi.org/10.1525/bio.2010.60.3.6
Simberloff, D., & Dayan, T. (1991). The guild concept and the structure of ecological communities. *Annual Review of Ecology and Systematics*, *22*, 115–143. http://dx.doi.org/10.1146/annurev.ecolsys.22.1.115
Sherry, T. W., & Holmes, R. T. (1985). Dispersion patterns and habitat responses of birds in northern hardwoods forests. In M. L. Cody (Ed.), *Habitat selection in birds* (pp. 283–309). Orlando: Academic Press.
Straus, M. A., Bavrlic, K., Nol, E., Burke, D. M., & Elliott, K. A. (2011). Reproductive success of cavity-nesting birds in partially harvested woodlots. *Canadian Journal of Forest Research*, *41*(5), 1004–1017. http://dx.doi.org/10.1139/X11-012
Strohbach, M. W., Haase, D., & Kabisch, N. (2009). Birds and the city: Urban biodiversity, land use, and socioeconomics. *Ecology and Society*, *14*(2).
Tirpak, J. M., Jones-Farrand, D., Thompson, F. R., III, Twedt, D. J., Baxter, C. K., Fitzgerald, J. A., et al. (2009). Assessing ecoregional-scale habitat suitability index models for priority landbirds. *Journal of Wildlife Management*, *73*(8), 1307–1315.
Tirpak, J. M., Jones-Farrand, D. T., Thompson, F. R., III, Twedt, D. J., & Uihlein, W. B., III. (2009). *Multiscale habitat suitability index models for priority landbirds in the Central Hardwoods and West Gulf Coastal Plain/Ouachitas Bird Conservation Regions (General Technical Report NRS-49)*. Newtown Square, PA: U.S.

Department of Agriculture, Forest Service, Northern Research Station, USA. Retrieved from:. http://www.nrs.fs.fed.us/pubs/9723i

U.S. Census Bureau. (2012). *Growth in urban populations outpaces rest of nation, Census Bureau reports*. Retrieved from:. http://www.census.gov/newsroom/releases/archives/2010_census/cb12-50.html

U.S. Fish and Wildlife Service. (2012). *Urban conservation treaty for migratory birds*. Retrieved from:. http://www.fws.gov/migratorybirds/Partnerships/UrbanTreaty/urbantreaty.html

Warren, P. S., Ryan, R. L., Lerman, S. B., & Tooke, K. A. (2011). Social and institutional factors associated with land use and forest conservation along two urban gradients in Massachusetts. *Landscape and Urban Planning*, *102*(2), 82–92. http://dx.doi.org/10.1016/j.landurbplan.2011.03.012

Watts, B. D. (1999). *Partners in flight: Mid-Atlantic Coastal Plain bird conservation plan: physiographic Area #44*. Retrieved from:. http://www.partnersinflight.org/bcps/plan/pl_44_10.pdf

Supplementary Material
Species Accounts and Models

American Robin

Status

The American Robin *Turdus migratorius* is one of North America's most abundant, widespread and recognizable birds. This familiar migratory species thrives in both suburban and wildland settings, and has been deemed "America's favorite songbird" (Sharp, 1990). American Robin populations are either stable or increasing throughout the range; deforestation, urbanization and agricultural development often create habitat for the species (Sallabanks & James, 1999). Due to its ability to thrive in a variety of habitat conditions, the American Robin does not have a conservation status. However, it is used as a bioindicator for chemical pollution. The species has enjoyed a 1.9 percent increase between 1966-1996 (Sallabanks & James, 1999).

Natural History

A lower canopy, shrub forager, the American robin's diet varies seasonally between earthworms and other soft invertebrates in spring and summer, and fruit in fall and winter (Sallabanks & James, 1999). Foraging substrates include lawns, loamy soil, and fruit bearing trees, shrubs and vines. Breeding habitat ranges from open woodlands and woodland edges and clearings, fields, orchards, and shade trees in residential areas. Residential areas and parks with lawns interspersed with shrubs and trees are ideal. Nesting sites vary and include horizontal branches or forks of a tree, shrubs and ledges of buildings. The sky blue eggs and speckled nestlings are familiar to many suburbanites. Territory sizes vary with population density, ranging between 0.04 and 0.84 ha (Pitts, 1984; Young, 1950). Winter territorial behavior focuses around the defense of fruit (Young, 1950).

Model Description

The habitat suitability index (HSI) model for the American robin includes two plot variables: percent canopy cover and percent lawn cover as estimated for a 0.04 ha plot. Although forest area has been shown to limit robin populations (Keller, Robbins, & Hatfield, 1993; Robbins, Dawson, & Dowell, 1989), we believe that local features are better predictors for this species.

We based our assumed values for canopy percent on qualitative accounts of the species requiring some trees yet not requiring extensive woodlands (Table 1). We fit a rational function (0.6439+(-0.0024*Canopy Percent))/(1+(-0.0312*Canopy Percent)+(0.0005*Canopy Percent^2)) through these data points to quantify the relationship between canopy coverage and the suitability index (SI score; Fig. 1).

We based our assumed values for lawn percent on qualitative accounts and personal observations of the species extracting earthworms and other soft invertebrates from manicured lawns in wooded parks and residential yards (Table 2). Since the robin primarily nests in trees and shrubs, an area with 100 % lawn cover would not be suitable. The relationship reflects the inverse of percent canopy. We fit a reciprocal quadratic function (1/(4.1918+(-0.0831*Lawn

Percent)+(0.00051* Lawn Percent ^2)) through these data points to quantify the relationship between lawn coverage and the suitability index (SI score; Fig. 2). We calculated the geometric mean of these habitat models to generate a final SI score for this species.

Baltimore Oriole

Status

The Baltimore oriole (*Icterus galbula*) is a long-distance neotropical migrant found throughout north eastern and central United States, and the plains of Canada. This species has adapted well to suburbia and urban parks, and thus, is another of America's most familiar songsters. The management of treed parks in urban and suburban areas will assist with the broadening of the breeding distribution (Ickes, 1992). The species has a Partners in Flight (PIF) score of 17 (a score of 30 being the highest for this region and thus having the highest level of conservation concern) in the mid-Atlantic region and a 3.2 percent decline between 1966-1996 (Watts, 1999). The PIF score in southern New England is 23 (Dettmers & Rosenberg, 2000).

Natural History

This canopy-gleaning passerine is found in a variety of habitats, favoring deciduous woodland edges, especially along riparian corridors, and suburban areas with tall and scattered shade trees, groves, orchards and parks. Also found in open woodlands with well-spaced trees (Salt & Salt, 1976); avoids closed-canopy forests (Palmer-Ball, 1996), and prefers large trees (e.g. dbh > 23 cm; Perkins, Johnson, & Blankenship, 2003). The species nests in deciduous trees, and builds their pendulant nest in the upper canopy, near the tip or outer branch of a tree (Rising & Flood, 1988). The familiar pendulant nest droops down from the upper branches. Territory sizes range from 0.15 ha to 1.86 ha.

Model Description

The HSI for Baltimore oriole includes one plot variable: canopy percent within a 0.04 ha plot and one tree variable: tree density for trees greater than 23 cm diameter at breast height. These variables address the relationship between a moderate canopy cover that consists of primarily large, open-grown trees.

We based our assumed values for canopy percent on qualitative accounts of the species requiring some trees yet not requiring extensive woodlands (Table 3). We fit a gausian function 1.0127*exp(0-((Canopy Percent-35.4635)^2)/(2*15.3508^2)) through these data points to quantify the relationship between canopy coverage and the suitability index (SI score; Fig. 3).

We based our assumed values for the upper limits of large tree density (>23 cm dbh) per 0.04 ha plot on i-Tree data sets: four of the ten cities had plots with at least 11 large trees present (Table 4). In addition, a higher density of large trees would increase the canopy coverage for the plot. The tree size also positively correlates with tree height and therefore larger trees are taller and thus more suitable for this high-canopy nester. We fit a rational function (0.0378+(0.2794*large tree density))/(1+(-0.4471* large tree density)+(0.1311* large tree density ^2))through these data points to quantify the relationship between large tree density and the suitability index (SI score; Fig. 4). We calculated the geometric mean of these habitat models to generate a final SI score for this species.

Black-capped Chickadee
Carolina Chickadee

Status

The black-capped chickadee (*Parus atricapillus*) is one of America's most widespread and familiar species. This non-migratory species can be found throughout the northern half of the United States and much of Canada. Based on Breeding Bird Surveys, eastern populations are thought to be increasing, though the range expansion of tufted titmouse (*Baeolophus bicolor*) might negatively impact chickadee populations (Loery & Nichols, 1985; Smith, 1991). Although urbanization has a negative effect on black-capped chickadees, suburban areas with large natural snags seem to partially mitigate the impacts of urban development (Blewett & Marzluff, 2005; Donnelly & Marzluff, 2006). The species has a PIF score of 15 in the mid-Atlantic region (Watts, 1999). No PIF scores reported for southern New England (Dettmers & Rosenberg, 2000).

The Carolina chickadee (*Parus carolinensis*) is the southeastern counterpart of black-cappped chickadee, with western limits in Kansas and eastern Texas and northern reaches into New Jersey and Pennsylvania. Similar to black-capped chickadees, the Carolina chickadee has adapted to suburbanization due to the presence of bird feeders and nest boxes (Doherty & Grubb 2002a; Hadidian, Sauer, Swarth, Handly, Droege, Williams, Huff, & Didden, 1997; Ringler, 1996). However, suburban areas highly prone to habitat fragmentation and areas with strong, negative interactions with house wrens (*Troglodytes aedon*) might lead to negative population trends (Doherty & Grubb, 2002b; Foote, Mennill, Ratcliffe, & Smith, 2010; Mostrom, Curry, & Lohr, 2002). Carolina chickadee has a PIF score of 21 and a 2.2 percent decline between 1966 and 1996 in the mid Atlantic region (Watts, 1999).

Natural History

A lower canopy, shrub gleaner, both the black-capped and Carolina chickadee diet consists mainly of insects during the breeding season and a mixture of seeds and berries, and insects and spiders during the winter (Smith, 1991; 1993). Breeding habitat includes deciduous, coniferous, or mixed woodlands (mixed preferred for black-capped, deciduous preferred for Carolina; Morse, 1970), and both species can be found in heavily forested and residential areas, with optimal conditions of an open understory and mature subcanopy (Anderson & Shugart, 1974). Wintering habitat includes city parks and residential areas with feeding stations adjacent to breeding habitat. Specific habitat requirements include dead standing trees or stubs (minimum dbh 10 cm; Holmes, 2002) for excavating cavities or trees with existing cavities for nesting (Mostrom, Curry, & Lohr, 2002). The chickadees will also use nest boxes.

Model Description

The US Fish and Wildlife Service developed an HSI model for black-capped chickadee which included percent canopy cover (i-Tree plot variable), tree height and dead wood density (i-Tree

tree variable; Holmes, 2002; Schroeder, 1982). We used the revised models published by Holmes (2002) for the i-Tree datasets. Due to the similarities in habitat requirement (Mostrom, Curry, & Lohr, 2002), we assumed that the variables thought to limit population abundances of black-capped chickadee was applicable to Carolina Chickadee and therefore present one model for both species.

Sturman (1968) proposed that tree foliage volume was a strong predictor of arthropod abundance, a major food resource for chickadees. However, this is time consuming to measure, and therefore Schroeder (1983) suggested that canopy cover and mean tree height per 0.04 ha plot were suitable alternative variables to address food resources (Table 5, 6). Although these data are available in i-Tree, we stuck with Schroeder's models. We fit a gausian function (1.002*exp((0-((Canopy Percent)-63.5682)^2)/1795) through these data points to quantify the relationship between canopy coverage and the suitability index (SI score; Fig. 5). We fit a logistic function (0.9757/(1+(11.7426*exp(-0.4852*Mean Tree Ht(m)))) through the data points to quantify the relationship between mean tree height (m) per 0.04 ha plot and the suitability index (SI score; Fig. 6).

To address nesting resources we calculated the number of trees within a 0.04 ha plot with deadwood. To calculate the density of dead wood per i-Tree plot, we assumed all trees with a condition classification of "fair", "poor", "dying" and "dead" potentially harbored wood that could be excavated for cavities. In addition to Holmes' (2002) values, we included Sedgwick and Knopf (1990) data that sites with at least six trees with dead wood were most suitable (Table 7). We fit a logistic function (1.007/(1+(32.567*exp(-1.403*density of trees with dead wood))) through the data points to quantify the relationship between density of trees with deadwood per 0.04 ha plot and the suitability index (SI score; Fig. 7). We calculated the geometric mean of these habitat models to generate a final SI score for this species.

European Starling

Status

From humble beginnings of about 100 individuals released in Central Park, NY, the European Starling (*Sturnus vulgaris*) is one of the most successful introductions to North America with a population hovering around 200 million (Cabe, 1993). Although declining in Europe (Newton, 2004), this species continues to thrive throughout North American cities, towns and agricultural areas. Due to its overabundance, this species is considered a nuisance and efforts to control populations are common, though often ineffective. Starlings often compete with native species for cavity sites and thus their increasing presence has detrimental impacts on other cavity-nesting species (Cabe, 1993). The species has a PIF score of 11 and a 2.7 decline between 1966-1996 in the mid-Atlantic region (Watts, 1999). No conservation status provided for southern New England (Dettmers & Rosenberg, 2000).

Natural History

A ground-foraging omnivorous passerine, the European starling is one of North America's most synanthropic species. Urban (e.g. lawns) and cultivated fields and hayfields, orchards, and parks provide ideal conditions for this species and can often be seen foraging for insects, grains, fruit and seeds in these areas. They appear to avoid pristine wilderness areas including non-grasslands, forests, deserts and arid chaparral (Cabe, 1993). They form massive winter roosts in dense vegetation, with numbers exceeding a million individuals. The shimmering flight pattern of a tight flock is a common spectacle over fields and cities alike. This cavity nester will use a variety of holes including crevices in buildings, cliffs, nest boxes and previously occupied woodpecker cavities. The territories are focused within the immediate vicinity of the cavity entrance (ca. 50 cm; Kessel, 1957).

Model Description

The habitat suitability index (HSI) model for the European starling includes two plot variables: percent lawn cover and percent building cover as estimated for a 0.04 ha plot, and two tree variables: tree density and density of deadwood present within a 0.04 ha plot. Although forest area has been shown to limit starling populations (Keller, Robbins, & Hatfield, 1993; Robbins, Dawson, & Dowell, 1989), we believe that local features are better predictors for this species.

We based our assumed values for lawn percent on qualitative accounts and personal observations of the species extracting insects and seed from manicured lawns in wooded parks and residential yards to areas with extensive amounts of lawn and turf (Table 8). We fit a logistic function (1.02247436719/(1+(40.643183849*exp(-0.1043766533*lawn percent per 0.04 ha))) through these data points to quantify the relationship between lawn coverage and the suitability index (SI score; Fig. 8).

We based our assumed values for building percent on qualitative accounts and personal observations of this species nesting in rain gutters, eaves, and other building cavities. Since the starling is extremely synanthropic, we assumed that plots with 100% building cover were suitable despite the presence of lawn (Table 9). We fit a rational function (-0.0004+(0.0148*building percent per 0.04 ha))/(1+(-0.0379* building percent per 0.04 ha)+(0.0007* building percent per 0.04 ha ^2)) through these data points to quantify the relationship between building coverage and the suitability index (SI score; Fig. 9).

We based our assumed values for tree density per 0.04 ha to reflect a gradient from field to forest, with areas of low tree density (all size classes) being more suitable (Table 10). In addition, tree density reflects the inverse relationship with lawn percent. We fit a rational function (0.8129+(-0.088*tree density per 0.04 ha))/(1+(-0.3167* tree density per 0.04 ha)+(0.0547* tree density per 0.04 ha ^2)) through these data points to quantify the relationship between tree density and the suitability index (SI score; Fig. 10).

European starling also nests in cavities and we assumed some conditions suitable for woodpeckers (e.g. red-bellied woodpecker) would also apply for starlings. Straus, Bavrlic, Nol, Burke, and Elliott, (2011) found that when at least one tree with deadwood was present within a 0.04 ha plot, red-bellied woodpeckers were also present. Adkins, Giese, and Cuthbert (2003) found slightly higher densities of deadwood to be more suitable. Due to starlings' ability to nest in artificial cavities (i.e. not tree cavities), we assumed that sites without deadwood present were still suitable (Table 11). We fit an exponential function (0.8005*(1.2498-exp(-2.4290*density of trees with deadwood)) through these data points to quantify the relationship between deadwood density within 0.04 ha and the suitability index (SI score; Fig. 11). We then calculated the geometric mean of these habitat models to generate a final SI score for this species.

Northern Cardinal *Cardinalis cardinalis*

Status

The Northern Cardinal (*Cardinalis cardinalis*) is resident found throughout eastern and central United States and Mexico. Northward movements of its range have been attributed to the urban heat island effect and provisions at bird feeders. The brilliantly red male is one of America's most familiar birds and is the state bird for seven states (Halkin & Linville, 1999). Alteration of habitat (converting forests to agriculture and suburbs) has benefited cardinals by increasing nesting habitat (Halkin & Linville, 1999). Based on Breeding Bird Survey data, the species has experienced 1 percent decline between 1966-1996 and has a PIF score of 14 in the Mid-Atlantic region. The species has experienced a 3.3 percent increase between 1966 and 1996 (Watts, 1999), and no PIF score for southern New England (Dettmers & Rosenberg, 2000).

Natural History

This omnivorous ground gleaner's diet consists of seeds, fruit, and insects (Halkin & Linville, 1999). Bird feeding during winter assists with survival in the colder parts of its range. Cardinals can be found along forest edges, open woodlands, suburban yards, urban parks and other areas with thickets and shrubs. In an extensive study of a breeding bird community in east Tennessee, Anderson and Shugart (1974) found that cardinals preferred sites with a thick subcanopy and relatively open canopy. Nest sites are located in dense, low vegetation including shrubs and small trees (deciduous and coniferous), vines, thickets and briars (Conner, Anderson, & Dickson, 1986; Ehrhart & Conner, 1986), and with prominent song posts in close proximity (Dow, 1969). Territory size ranges from 0.21 to 2.60 ha (Halkin & Linville, 1999).

Model Description

The habitat suitability model (HSI) for northern cardinal includes two plot variables: canopy percent and shrub percent, both within a 0.04 ha plot. These variables address the relationship between a moderate to open canopy cover which can encourage an extensive shrub layer.

We based our assumed values for canopy cover on qualitative accounts and personal observations of the species in edge habitats, residential yards with little to no canopy cover and lack of observations in thick, extensive woodlands (Table 12). We fit a rational function (0.6313+(-0.0054* Canopy Percent))/(1+(-0.0370* Canopy Percent)+(0.0007* Canopy Percent ^2)) through the data points to quantify the relationship between percent canopy cover per 0.04 ha plot and the suitability index (SI score; Fig. 12).

We based our assumed values for shrub cover on qualitative accounts and personal observations of the species nesting in dense shrubs, privets, thickets and other low vegetation in residential yards and urban parks (Table 13). We fit a rational function (0.0095+(0.0213*Shrub Percent))/(1+(-0.0212*Shrub Percent)+(0.0004*Shrub Percent ^2)) through the data points to quantify the relationship between percent shrub cover per 0.04 ha plot and the suitability index (SI score; Fig. 13).

Red-bellied Woodpecker

Status

The red-bellied woodpecker (*Melanerpes carolinus*) has a broad distribution throughout the eastern half of the United States. The species is resident throughout the eastern part of its range though northern birds move south during cold winters (Winkler, Christie, & Nurney, 1995). The red-bellied woodpecker is a familiar site to feeder watchers and easily recognized. The species is commonly reported throughout the eleven i-Tree cities and not of conservation concern largely in part to its preference for a wide range of forest types (Shackelford, Brown, & Conner, 2000). Based on Breeding Bird Survey data, the nationwide population is either stable or increasing (Price, Droege, Price, & Beadle, 1995) and appears to thrive in urban and suburban areas. However, similar to other woodpeckers, the red-bellied is heavily dependent on snags and dead wood for nesting and roosting and therefore urban forest management plans that encourage dead wood have the potential to support this species. For the mid-Atlantic region, the species has a PIF score of 15 (Watts, 1999), and in southern New England, the species has increased 21.1 percent between 1966 and 1996 (Dettmers & Rosenberg, 2000).

Natural History

The red-bellied woodpecker is a vocal and conspicuous cavity-nester found in mature pine forests, hardwood forests or a mixture of the two. The species excavates cavities in snags on dead trees or dead limbs on live trees. An opportunistic forager, this species' diet consists of fruit, beech and acorn masts and arboreal arthropods. Red-bellied woodpeckers are sedentary, remaining on breeding grounds year-round. Average territory size ranges from 1.8 to 2.5 ha based on studies from upland forests and virgin floodplain forest in Illinois (Shackelford, Brown, & Conner, 2000).

Model Description

The habitat suitability index model for the red-bellied woodpecker includes four plot variables: tree density per 0.04 ha, basal area per ha, density of dead wood (i.e. trees classified as fair, poor, dying or dead) per 0.04 ha, and percent canopy cover per 0.04 ha.

The species relies on forested areas and we included three variables to describe these habitat needs. Adkins, Giese, and Cuthbert (2003) observed 24 trees per 0.04 ha and a basal area of 34 m^2 per ha in oak forests of the Upper Midwest, while Conner (1980) observed 30 trees per 0.04 ha and a basal area of 14 m^2 per ha in oak-hickory forests around Blacksburg, VA. However, these studies didn't discern tree size. We wanted the model to reflect the mean diameter of the cavity limb (21.6 cm; Jackson, 1976) so only included trees greater than 23 cm dbh and adjusted the densities to reflect these conditions (Table 14). We fit a rational function (0-0.0035+(0.1606*Tree Density))/(1+(-0.1417*Tree Density)+(0.0233* Tree Density ^2)) where

Tree Density is the density of trees greater than 23 cm dbh within a 0.04 ha plot, through these data points to predict how habitat suitability varied with large tree density (Fig. 14). We assumed suitability was the lowest when trees were absent. Our inclusion of basal area for all trees greater than 6 cm dbh reflects the propensity for this species to prefer relatively dense forests (Shackelford, Brown, & Conner, 2000; Table 15). We fit a logistic function 0.9906/(1+(47.9216*exp(-0.9689*basal area))) where basal area is m^2/ha and calculated for all trees greater than 6 cm dbh, through these data points to quantify the relationship between basal area and the suitability index (SI score; Fig. 15).

Canopy coverage also predicts habitat suitability. DeGraaf, Yamasaki, Leak, and Lester, (2006) suggested that when canopy coverage exceeding 35%, the site provided suitable conditions for red-bellied woodpeckers. We based our assumed values for canopy cover on qualitative accounts and personal observations of the species in forested suburban and riparian areas, with lack of observations in areas with little to no canopy cover and areas with an extremely dense canopy cover (Table 16). We fit a rational function (-0.0371+(0.0124*Percent Canopy))/(1+(-0.0363* Percent Canopy)+(0.0005* Percent Canopy ^2)), where Percent Canopy is the percent of a 0.04 ha plot with tree canopy cover, through these data points to predict how habitat suitability varied with canopy coverage (Fig. 16). We assumed suitability was the lowest when trees were absent.

Although dead wood is necessary for foraging and nesting, they are not essential for detecting red-bellied woodpeckers. Of 42 nests in southwest Ontario, Strauss et al. (2011) observed 93% of the nests in dead and declining trees and 6% of nests in healthy trees. Adkins, Giesse, and Cuthbert (2003) observed 3 dead or declining trees per 0.04 ha in the Midwest (Table 17). We fit a logistic function 1/(1+(15.67*exp(-5.338*Dead Wood density per 0.04 ha))), (where dead wood is recorded as trees with a condition of fair, poor, dying or dead) through these data points to quantify the relationship between trees with dead wood and the suitability index (SI score; Fig. 17). We calculated the geometric mean of these habitat models to generate a final SI score for this species.

Scarlet Tanager

Status

The scarlet tanager (Piranga olivacea) is a long-distance neotropical migrant, found in deciduous forests throughout the northeastern United States and southern Canada. This forest interior species is highly sensitive to forest fragmentation (Roberts & Norment, 1999). In a study from New Jersey, scarlet tanagers were present only in forest fragments greater than 3 ha, though forest areas greater than 10 ha were required to sustain a viable population (Galli, Leck, & Forman, 1976; Robbins, Dawson, & Dowell, 1989; Roberts & Norment, 1999). As fragment size decreases, nest predation and parasitism rates increase (Robinson, Thompson III, Donovan, Whitehead, & Faaborg, 1995). According to the Breeding Bird Survey, the species has experienced a1.6 percent decline between 1966 and1996, and has a PIF score of 21 in the mid-Atlantic region (Watts, 1999). The species has experienced a 1 percent decline and has a PIF score of 22 in southern New England (Dettmers & Rosenberg, 2000).

Natural History

Scarlet tanagers spend most of their time in the mid to upper canopy, hovering and gleaning insects from flowers, fruit, leaves and bark (Mowbray, 1999). They are associated with mature deciduous and mixed forests but occasionally found in dense shade trees in suburban areas, cemeteries and parks (Mowbray, 1999). They prefer trees greater than 22.4 cm dbh, and primarily in oak-hickory woods (Mowbray, 1999). Territory size varies according to vegetation type but ranges from 0.8 – 5.0 ha (Robbins, 1980; Zumeta & Holmes, 1978).

Model Description

The HSI model for the scarlet tanager includes three plot variables: large tree (> 23 cm dbh) density per 0.04 ha, basal area per ha, and percent canopy cover per 0.04 ha. The model also includes one landscape variable: extent of forest patches (ha).

Three separate studies found that when a forested 0.04 ha plot had at least 20 large trees, the area would support scarlet tanagers (Shy, 1984; Roberts & Norment, 1999; Rivera, McShea, & Rappole, 2003; Table 18). We fit a logistic function (1.01622702/(1+(24569.22035*EXP(-0.6493929*Tree Density))) where Tree Density is the density of trees greater than 23 cm dbh within a 0.04 ha plot, through these data points to predict how habitat suitability varied with large tree density (Fig. 18). We assumed suitability was the lowest in plots with fewer than ten trees. In addition to mature forests (i.e. those with large trees), scarlet tanagers also prefer dense forests. Based on Roberts and Norment (1999) we calculated a mean basal area of 62 m^2/ha and assumed this density to be most suitable for tanagers (Table 19). We fit a logistic function (1.0363/(1+(49.295*EXP(-0.1088*Basal Area))) where basal area is m^2/ha and calculated for all trees greater than 6 cm dbh, through these data points to quantify the relationship between basal area and the suitability index (SI score; Fig. 19).

Scarlet tanagers prefer territories in forested areas with dense canopy cover (Ambuel & Temple, 1983). Roberts and Norment (1999) and Shy (1984) suggest that a canopy coverage of 89% represented the most suitable conditions for this species, with forested patches having 75% canopy coverage highly suitable (Shy, 1984; Table 20). We fit a logistic function (1.0363/(1+(49.295*EXP(-0.1088*Percent Canopy))) where Percent Canopy is the percent of a 0.04 ha plot with tree canopy cover, through these data points to predict how habitat suitability varied with canopy coverage (Fig. 20). We assumed suitability declined significantly when canopy coverage was less than 50%.

We included one landscape variable to account for the extreme sensitivity this species exhibits to forest fragmentation (Robinson et al., 1995; Table 21). We fit a multiple multiplicative factor (MMF) function ((-0.0009*4.3992)+(1.6780*Forest Area ^0.2539))/(4.3992+ Forest Area ^0.2539) where Forest Area is forest patch size (ha) through these data points to predict how habitat suitability varied with the extent of forested areas (Fig. 21). Although a low suitability, forest patches 1 ha have the potential to harbor scarlet tanagers, thus suggesting the possibility of urban remnant patches to have some conservation value for this species. We calculated the geometric mean of these habitat models to generate a final SI score for this species.

Wood Thrush

Status

The wood thrush (Hylocichla mustelina) is a long-distance neotropical migrant found throughout eastern North America (Roth, Johnson, & Underwood, 1996). This species is a symbol of threatened and declining songbirds and it has become increasingly rare throughout its range since the 1970s (Evans, Gow, Roth, Johnson, & Underwood, 2011). Habitat loss and fragmentation in the breeding and wintering grounds have had detrimental effects on populations, and thus exacerbating the impacts of cowbird parasitism (Thompson III, Robinson, Donovan, Faaborg, Whitehead, & Larsen, 2000). The wood thrush is a species of conservation concern, with a 2.3 percent decline between 1966 and 1996, and a Partners in Flight score of 24 for the mid Atlantic (Watts, 1999). The species has experienced a 2.2 percent decline between 1966 and 1996, and PIF score of 24 for southern New England (Dettmers & Rosenberg, 2000).

Natural History

A ground-foraging passerine, the wood thrush is associated with mature upland forests (mainly deciduous or mixed but largely avoids evergreen stands) with closed overstory canopies (Bell & Whitmore, 2000; Evans et al., 2011). Additional conditions include forest patches with trees taller than >16 m, moderate subcanopy, sapling density and shrubs for nesting, cool and moist soil conditions, and a somewhat open forest floor with decaying leaf litter for foraging (Evans et al., 2011). The species is thought to be highly sensitive to forest fragmentation with regards to its productivity but has nested in small forest fragments (0.3 ha; e.g. remnant patches in residential areas and parks) at low densities (Tilghman, 1987; Weinberg & Roth, 1998).

The wood thrush's diet mainly consists of soil invertebrates and fruits, and occasionally feeds on arboreal insects, snails and small salamanders (Evans et al., 2011). The nest is typically located on a horizontal branch or crotch within a sapling or tree (Evans et al., 2011). Hoover and Brittingham (1998) suggested that nest success was better predicted by the amount of forest in the landscape rather than the microhabitat structural features surrounds nests. Territory size ranges between 0.08 and 4.0 ha (Evans, Stutchbury, & Woolfenden, 2008; Twomey, 1945).

Habitat Model

The HSI model for the wood thrush includes two plot variables: percent canopy cover per 0.04 ha and sapling density per 0.04 ha plot. The model also includes one landscape variable: percent forest landcover within a 1 km radius.

The wood thrush associates with dense canopied forests (Table 22) and we fit a logistic function 1.03163/(1+(141241.64*EXP(-0.1531*Percent Canopy))) where Percent is the percent of a 0.04 ha plot with tree canopy cover to data from Annand and Thompson (1997) and Hoover and Brittingham (1998) to predict SI scores from percent canopy coverage scores (Fig. 22). Tirpak et al. (2009) devised a model that incorporated small stem densities (< 2.5 cm) based on Hoover and Brittingham (1998) assertion that 1,988 stems per ha represented optimal habitat. These stem densities within the i-Tree datasets were far from this abundance, even when we included shrub densities into the equation. We therefore used sapling density (<10 cm) as a proxy to assess the midstory cover. We based our assumed values on existing i-Tree datasets and balanced the relationship between a dense canopy cover and the sapling density whereby at least 10 saplings were recorded in i-Tree plots with 90% canopy coverage (Table 23). We fit a logistic function (1.0401978/(1+(65.800186*EXP(-0.758149*(Sapling Density))))), where sapling density is the number of trees <10 cm in diameter within a 0.04 ha plot, through these data points to quantify the relationship between sapling density and the suitability index (Fig. 23).

Although studies have demonstrated the importance of forest area as a predictor of nest success for wood thrush (e.g. Robbins, Dawson, & Dowell, 1989; Kilgo et al., 1998), we chose a variable that reflected the amount of forest patches within the greater landscape matrix to address the suitability of urban parks and remnant patches within residential landscapes (Table 24). Following Tirpak et al. (2008), we fit a logistic function 1.003/(1+(224.7853*EXP(-0.1081*(1KM % Forest)))) where 1KM % Forest is the percent of a 1 km plot around an i-Tree monitoring plot classified as forest landcover to data based on Donovan, Jones, Annand, and Thompson (1997; Fig. 24). In this study, the predator and brood parasite communities were related to fragmentation size: highly fragmented (< 15 percent), moderately fragmented (45 to 50 percent), and lightly fragmented (> 90 percent forest) landscapes. We followed logic from Tirpak et al. (2008), and also assumed the midpoints between 30 and 70 percent represented the thresholds for low suitability (SI score ≤ 0.10) and excellent suitability (SI score ≥ 0.90) habitats. We calculated the geometric mean of these habitat models to generate a final SI score for this species.

Literature Cited

Adkins Giese, C. L., & Cuthbert, F. J. (2005). Woodpecker nest tree characteristics in upper midwestern oak forests. *The Canadian Field-Naturalist*, 119(3), 367–376.

Ambuel, B., & Temple, S. A. (1983). Area-Dependent Changes in the Bird Communities and Vegetation of Southern Wisconsin Forests. *Ecology*, 64(5), 1057–1068. doi:10.2307/1937814

Anderson, S., & Shugart, H. (1974). Habitat selection of breeding birds in an east Tennessee deciduous forest. *Ecology*, 55(4), 828–837. doi:10.2307/1934418

Annand, E. M., & Thompson, F. R. (1997). Forest Bird Response to Regeneration Practices in Central Hardwood Forests. *The Journal of Wildlife Management*, 61(1), 159–171. doi:10.2307/3802425

Bell, J. L., & Whitmore, R. C. (2000). Bird Nesting Ecology in a Forest Defoliated by Gypsy Moths. *The Wilson Bulletin*, 112(4), 524–531.

Blewett, C. M., & Marzluff, J. M. (2005). Effects of Urban Sprawl on Snags and the Abundance and Productivity of Cavity-Nesting Birds. *The Condor*, 107(3), 678–693.

Cabe, P. R. (1993). European Starling (Sturnus vulgaris). (A. Poole, Ed.). *The Birds of North America Online*. Retrieved from http://bna.birds.cornell.edu/bna/species/048 doi:10.2173/bna.48

Conner, R. (1980). Foraging habits of woodpeckers in southwestern Virginia. *Journal of Field Ornithology*, 51(2), 119–127.

Conner, R. N., Anderson, M. E., & Dickson, J. G. (1986). Relationships among Territory Size, Habitat, Song, and Nesting Success of Northern Cardinals. *The Auk*, 103(1), 23–31.

DeGraaf, R. M., Yamasaki, M., Leak, W. B., & Lester, A. M. (2006). *Technical guide to forest wildlife habitat management in New England*. Burlington, Vermont: University of Vermont Press.

Dettmers, R., & Rosenberg K.V. (2000). *Partners In Flight landbird conservation plan: physiographic area 9: Southern New England.* Retrieved from http://www.partnersinflight.org/bcps/plan/pl_09_10.pdf.

Doherty, P. F., & Grubb, T. C. (2002a). Survivorship of Permanent-Resident Birds in a Fragmented Forested Landscape. *Ecology*, 83(3), 844–857. doi:10.1890/0012-9658(2002)083[0844:SOPRBI]2.0.CO;2

Doherty, P. F., & Grubb, T. C. (2002b). Nest usurpation is an "edge effect" for Carolina chickadees Poecile carolinensis. *Journal of Avian Biology*, 33(1), 77–82. doi:10.1034/j.1600-048X.2002.330112.x

Donnelly, R., & Marzluff, J. (2006). Relative importance of habitat quantity, structure, and spatial pattern to birds in urbanizing environments. *Urban Ecosystems*, 9(2), 99–117. doi:10.1007/s11252-006-7904-2

Donovan, T. M., Jones, P. W., Annand, E. M., & Thompson, F. R. (1997). Variation in Local-Scale Edge Effects: Mechanisms and Landscape Context. *Ecology*, 78(7), 2064–2075.

Dow, D. D. (1969). Habitat utilization by cardinals in central and peripheral breeding populations. *Canadian Journal of Zoology*, 47(3), 409–417. doi:10.1139/z69-075

Ehrhart, R. L., & Conner, R. N. (1986). Habitat selection by the Northern Cardinal in three eastern Texas forest stands. *The Southwestern Naturalist*, 31(2),191–199.

Evans, M. L., Stutchbury, B. J. M., & Woolfenden, B. E. (2008). Off-territory forays and genetic mating system of the Wood Thrush (Hylocichla mustelina). *The Auk*, 125(1), 67–75.

Evans, M., Gow, E., Roth, R.R., Johnson, M.S., & Underwood, T.J. (2011). Wood Thrush (Hylocichla mustelina). (A. Poole, Ed.). *The Birds of North America Online*. Retrieved from http://bna.birds.cornell.edu/bna/species/246 doi:10.2173/bna.246

Foote, J. R., Mennill, D.J., Ratcliffe, L.M. & Smith, S.M. (2010). Black-capped Chickadee (Poecile atricapillus). (A. Poole, Ed.). *The Birds of North America Online*. Retrieved from http://bna.birds.cornell.edu/bna/species/039 doi:10.2173/bna.39

Galli, A. E., Leck, C. F., & Forman, R. T. T. (1976). Avian Distribution Patterns in Forest Islands of Different Sizes in Central New Jersey. *The Auk*, 93(2), 356–364.

Hadidian, J., Sauer, J., Swarth, C., Handly, P., Droege, S., Williams, C., Huff, J., et al. (1997). A citywide breeding bird survey for Washington, D.C. *Urban Ecosystems*, 1(2), 87–102. doi:10.1023/A:1018563125184

Halkin, S. L., & Linville, S. U. (1999). Northern Cardinal (Cardinalis cardinalis). (A. Poole, Ed). *The Birds of North America Online*. Retrieved from http://bns.birds.cornell.edu/bna/species/440 doi:10.2173/bna.440

Holmes, D. (2002). Habitat evaluation procedures (HEP) report; Pend Oreille Wetlands Wildlife II project. Technical Report, Project No. 199106000. Retrieved from https://pisces.bpa.gov/release/documents/documentviewer.aspx?doc=00004008-2

Hoover, J. P., & Brittingham, M. C. (1998). Nest-Site Selection and Nesting Success of Wood Thrushes. *The Wilson Bulletin*, 110(3), 375–383.

Ickes, R. (1992). Northern Oriole, Icterus galbula. In Brauning, D.W. (Ed.), *Atlas of the breeding birds in Pennsylvania* (pp. 410–411). Pittsburgh, PA: University of Pittsburgh Press.

Jackson, J. A. (1976). A Comparison of Some Aspects of the Breeding Ecology of Red-Headed and Red-Bellied Woodpeckers in Kansas. *The Condor*, 78(1), 67–76. doi:10.2307/1366917

Keller, C. M. E., Robbins, C. S., & Hatfield, J. S. (1993). Avian communities in riparian forests of different widths in Maryland and Delaware. *Wetlands*, 13(2), 137–144.

Kessel, B. (1957). A Study of the Breeding Biology of the European Starling (Sturnus vulgaris L.) in North America. *American Midland Naturalist*, 58(2), 257–331. doi:10.2307/2422615

Kilgo, J. C., Sargent, R. A., Chapman, B. R., & Miller, K. V. (1998). Effect of stand width and adjacent habitat on breeding bird communities in bottomland hardwoods. *The Journal of wildlife management*, 62(1), 72–83.

Loery, G., & Nichols, J. D. (1985). Dynamics of a Black-capped Chickadee Population, 1958-1983. *Ecology*, 66(4), 1195–1203. doi:10.2307/1939172

Morse, D. H. (1970). Ecological Aspects of Some Mixed-Species Foraging Flocks of Birds. *Ecological Monographs*, 40(1), 119–168. doi:10.2307/1942443

Mostrom, A. M., Curry, R. L., & Lohr, B. (2002). Carolina Chickadee (Poecile carolinensis). (A. Poole, Ed). *The Birds of North America Online*. Retrieved from http://bna.birds.cornell.edu/bna/species/636 doi:10.2173/bna.636

Mowbray, T. B. (1999). Scarlet Tanager (Piranga olivacea). (A. Poole, Ed.). *The Birds of North America Online*. Retrieved from http://bna.birds.cornell.edu/bna/species/479 doi:10.2173/bna.479

Newton, I. (2004). The recent declines of farmland bird populations in Britain: an appraisal of causal factors and conservation actions. *Ibis*, 146(4), 579–600. doi:10.1111/j.1474-919X.2004.00375.x

Palmer-Ball Jr, B. L. (1996). *The Kentucky breeding bird atlas*. University Press of Kentucky.

Perkins, M. W., Ron J. Johnson, & Blankenship, E. E. (2003). Response of Riparian Avifauna to Percentage and Pattern of Woody Cover in an Agricultural Landscape. *Wildlife Society Bulletin*, 31(3), 642–660.

Pitts, T. D. (1984). Description of American Robin territories in northwest Tennessee. *Migrant*, 55, 1–6.

Price, J., Droege, S., Price, A., & Beadle, D. D. (1995). *The summer atlas of North American birds*. Academic Press San Diego.

Ringler, R. F. (1996). Carolina Chickadee. In C. S. Robbins (Ed.), *Atlas of the Breeding Birds of Maryland and the District of Columbia* (pp. 256–257). University of Pittsburgh Press.

Rising, J.D. & Flood, N. J. (1998). Baltimore Oriole (Icterus galbula), (A. Poole, Ed.). *The Birds of North America Online.* Retrieved from http://bna.birds.cornell.edu/bna/species/384 doi:10.2173/bna.384

Rivera, J. H. V., McShea, W. J., & Rappole, J. H. (2003). Comparison of breeding and postbreeding movements and habitat requirements for the Scarlet Tanager (Piranga olivacea) in Virginia. *Auk*, 120(3), 632–644.

Robbins, C. S. (1980). Effect of forest fragmentation on breeding bird populations in the piedmont of the mid-Atlantic region. *Atlantic Naturalist*, 33(1), 31–36.

Robbins, C. S., Dawson, D. K., & Dowell, B. A. (1989). Habitat area requirements of breeding forest birds of the middle Atlantic states. *Wildlife Monographs*, 3–34.

Roberts, C., & Norment, C. J. (1999). Effects of plot size and habitat characteristics on breeding success of Scarlet Tanagers. *The Auk*, 116(1), 73–82.

Robinson, S. K., Thompson III, F. R., Donovan, T. M., Whitehead, D. R., & Faaborg, J. (1995). Regional forest fragmentation and the nesting success of migratory birds. *Science*, 267(5206),1987–1987.

Roth, R. R., Johnson, M. S., & Underwood, T. J. (1996). Wood Thrush (Hylocichla mustelina). (A. Poole, Ed.). *The Birds of North America Online*. Retrieved from http://bna.birds.cornell.edu/bna/species/246 doi:10.2173/bna.246

Sallabanks, R. & James, F.C. (1999). American Robin (Turdus migratorius), *The Birds of North America Online*. (A. Poole, Ed.). Retrieved from http://bna.birds.cornell.edu/bna/species/462 doi:10.2173/bna.462

Salt, W. R., & Salt, J. R. (1976). *The birds of Alberta: with their ranges in Saskatchewan & Manitoba*. Hurtig.

Schroeder, R.L. (1983). *Habitat suitability index models: Black-capped chickadee.* U.S. Department of the Interior, Division of Biological Services, Research and Development, Fish and Wildlife Service, Washington, DC. Retrieved from http://www.nwrc.usgs.gov/wdb/pub/hsi/hsi-037.pdf

Sedgwick, J. A., & Knopf, F. L. (1990). Habitat Relationships and Nest Site Characteristics of Cavity-Nesting Birds in Cottonwood Floodplains. *The Journal of Wildlife Management*, 54(1), 112–124. doi:10.2307/3808910

Shackelford, C. E., Brown, R. E., & Conner, R. N. (2000). Red-bellied Woodpecker (Melanerpes carolinus). (A. Poole, Ed.). *The Birds of North America Online*. (Retrieved from http://bna.birds.cornell.edu/bna/species/500 doi:10.2173/bna.500

Sharp, M. H. (1990). America's songbird-species profile: American Robin (Turdus migratorius). *Wildbird*, 4, 22–27.

Shy, E. (1984). Habitat shift and geographical variation in North American tanagers (Thraupinae: Piranga). *Oecologia*, 63(2), 281–285.

Smith, S. M. (1991). *The Black-capped Chickadee: behavioral ecology and natural history*. Comstock Pub Assoc.

Smith, S. M. (1993). Black-capped Chickadee (Poecile atricapillus). (A. Poole, Ed.). *The Birds of North America Online*. Retrieved from http://bna.birds.cornell.edu/bna/species/039 doi:10.2173/bna.39

Straus, M. A., Bavrlic, K., Nol, E., Burke, D. M., & Elliott, K. A. (2011). Reproductive success of cavity-nesting birds in partially harvested woodlots. *Canadian Journal of Forest Research*, 41(5), 1004–1017. doi:10.1139/X11-012

Sturman, W. A. (1968). Description and analysis of breeding habitats of the chickadees, Parus atricapillus and P. rufescens. *Ecology*, 49(3), 418–431.

Thompson III, F. R., Robinson, S. K., Donovan, T. M., Faaborg, J. R., Whitehead, D. R., & Larsen, D. R. (2000). Biogeographic, landscape, and local factors affecting cowbîrd abundance and host parasitism levels. In Smith, J. N. M., Cook, T. L., Rothstein, S. I., Robinson, S. K., Sealy, S. G. (Eds.), *Ecology and Management of Cowbirds and Their Hosts: Studies in the Conservation of North American Passerine Birds* (pp. 271-279). Austin, TX- University of Texas.

Tirpak, J. M., Jones-Farrand, D. T., Thompson III, F. R., Twedt, D. J., & Uihlein III, W. B. (2009). *Multiscale habitat suitability index models for priority landbirds in the Central Hardwoods and West Gulf Coastal Plain/Ouachitas Bird Conservation Regions* (General Technical Report NRS-49). Newtown Square, PA, U.S. Department of Agriculture, Forest Service, Northern Research Station, USA. Retrieved from http://www.nrs.fs.fed.us/pubs/9723i

Tilghman, N. G. (1987). Characteristics of urban woodlands affecting breeding bird diversity and abundance. *Landscape and Urban Planning*, 14, 481–495.

Twomey, A. C. (1945). The bird population of an elm-maple forest with special reference to aspection, territorialism, and coactions. *Ecological Monographs*, 15(2), 173–205.

Watts, B.D. (1999). *Partners in Flight: Mid-Atlantic Coastal Plain bird conservation plan: physiographic Area #44*. Retrieved from http://www.partnersinflight.org/bcps/plan/pl_44_10.pdf.

Weinberg, H. J., & Roth, R. R. (1998). Forest area and habitat quality for nesting Wood Thrushes. *The Auk*, 115(4),879–889.

Winkler, H., Christie, D. A., & Nurney, D. (1995). Woodpeckers: an identification guide to the woodpeckers of the world. *Houghton Mifflin Company, New York*.

Young, H. F. (1950). *Territorial behavior in the eastern robin*. University of Wisconsin–Madison.

Zumeta, D. C., & Holmes, R. T. (1978). Habitat shift and roadside mortality of Scarlet Tanagers during a cold wet New England spring. *The Wilson Bulletin*, 90(4), 575–586.

Table 1. Relationship between canopy percent per 0.04 ha and suitability index (SI) for American robin (AMRO) habitat, and associated references.

Canopy Percent (per 0.04 ha)	SI Score (AMRO)	Reference
0	0.7	assumed value
10	0.8	assumed value
20	0.9	assumed value
25	1	assumed value
30	1	assumed value
35	1	assumed value
40	0.6	assumed value
50	0.4	assumed value
70	0.3	assumed value
90	0.2	assumed value
100	0.1	assumed value

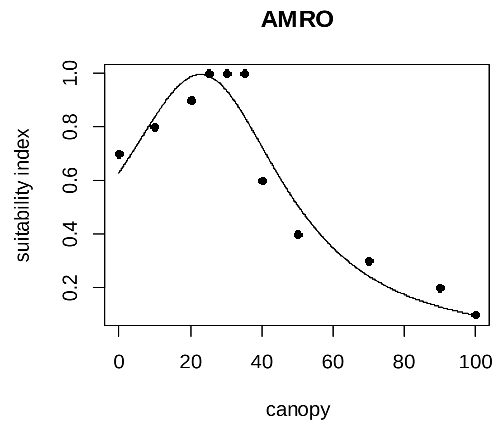


Table 2. Relationship between lawn percent per 0.04 ha and suitability index (SI) for American robin (AMRO) habitat, and associated references.

Lawn Percent (per 0.04 ha)	SI Score (AMRO)	Reference
0	0.2	assumed value
40	0.6	assumed value
80	1	assumed value
100	0.8	assumed value

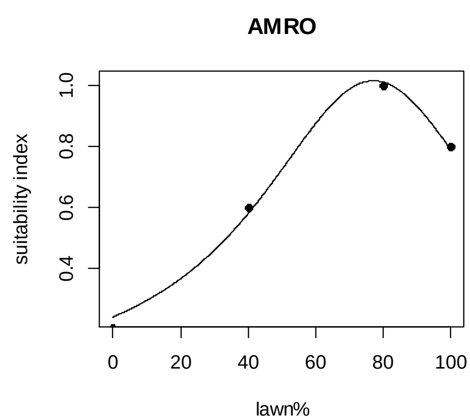


Table 3. Relationship between canopy percent per 0.04 ha and suitability index (SI) for Baltimore oriole (BAOR) habitat, and associated references.

Canopy Percent (per 0.04 ha)	**SI Score (BAOR)**	**Reference**
0	0	assumed value
10	0.3	assumed value
20	0.8	assumed value
30	1	DeGraaf et al., 2006
50	1	assumed value
70	0.2	Gotfryd, 1984
80	0.1	Palmer-Ball, 1996
100	0	assumed value

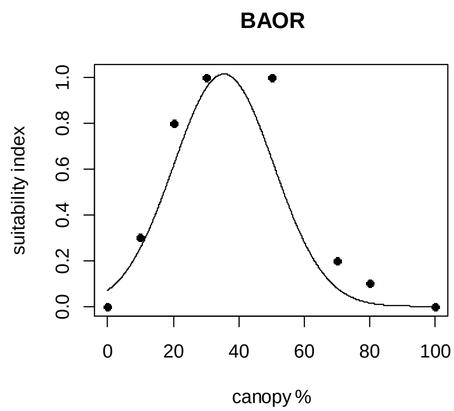


Table 4. Relationship between large tree density (dbh >23cm) and suitability index (SI) for Baltimore oriole (BAOR) habitat, and associated references.

Large Tree Density (per 0.04 ha)	**SI Score (BAOR)**	**Reference**
0	0	Perkins et al., 2003
1	0.5	assumed value
3	1	DeGraaf et al., 2006
5	0.7	assumed value
7	0.5	assumed value
9	0.3	assumed value
11	0.1	assumed value

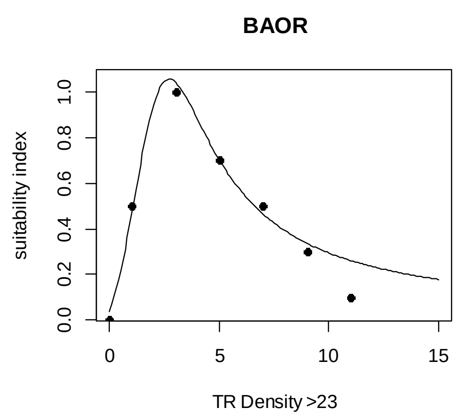


Table 5. Relationship between canopy percent per 0.04 ha and suitability index (SI) for black-capped chickadee (BCCH) habitat, and associated references. Carolina chickadee habitat models mimic black-capped chickadee.

Canopy Percent (per 0.04 ha)	SI Score (BCCH)	Reference
0	0	Holmes, 2002
12	0.25	Holmes, 2002
37	0.7	Holmes, 2002
50	1	Holmes, 2002
75	1	Holmes, 2002
87	0.8	Holmes, 2002

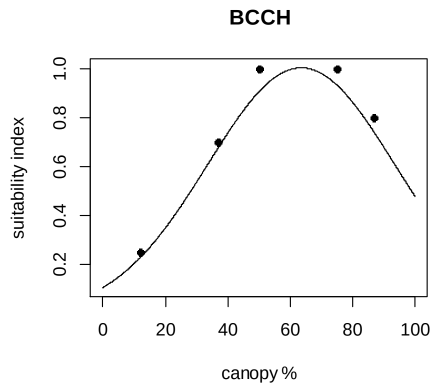


Table 6. Relationship between tree height (m) and suitability index (SI) for black-capped chickadee (BCCH) habitat, and associated references.

Mean Tree Height (m) (per 0.04 ha)	SI score (BCCH)	Reference
0	0	Holmes, 2002
1	0.15	Holmes, 2002
5	0.5	Holmes, 2002
10	0.85	Holmes, 2002
15	1	Holmes, 2002

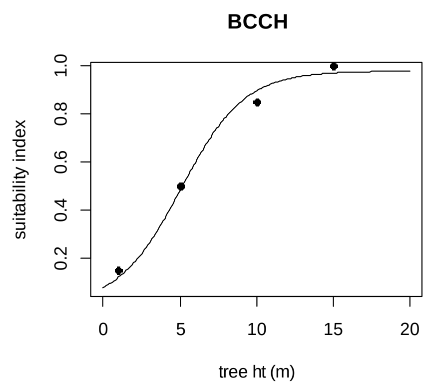


Table 7. Relationship between dead wood density per ha and suitability index (SI) for black-capped chickadee (BCCH) habitat, and associated references.

Dead Wood Density (per 0.04 ha)	SI Score (BCCH)	Reference
0	0.03	Holmes, 2002
4	0.9	Holmes, 2002
6	1	Sedgwick & Knopf, 1990

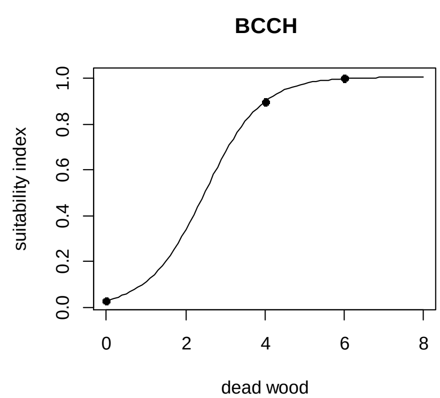


Table 8. Relationship between lawn percent per 0.04 ha and suitability index (SI) for European starling (EUST) habitat, and associated references.

Lawn Percent (per 0.04 ha)	SI Score (EUST)	Reference
0	0	assumed value
20	0.2	assumed value
40	0.6	assumed value
60	1	assumed value
80	1	assumed value
100	1	assumed value

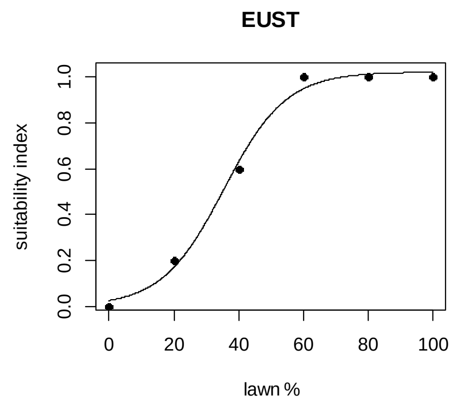


Table 9. Relationship between percent building and suitability index (SI) for European starling (EUST) habitat, and associated references.

Building Percent (per 0.04 ha)	SI Score (EUST)	Reference
0	0	assumed value
25	0.8	assumed value
50	1	assumed value
75	0.6	assumed value
100	0.4	assumed value

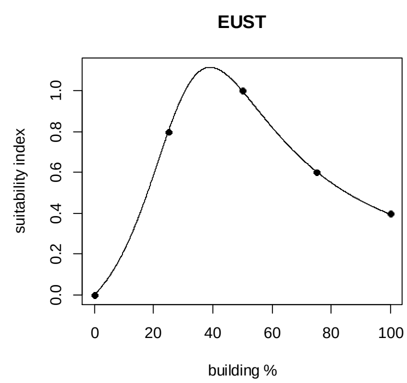


Table 10. Relationship between total tree density per 0.04 ha and suitability index (SI) for European starling (EUST) habitat, and associated references.

Tree Density (per 0.04 ha)	SI Score (EUST)	Reference
0	0.8	assumed value
1	1	assumed value
3	1	assumed value
5	0.5	assumed value
7	0.1	assumed value
10	0	assumed value

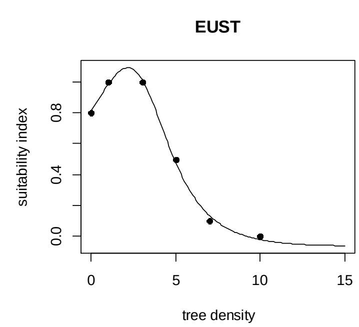


Table 11. Relationship between dead wood density per ha and suitability index (SI) for European starling (EUST) habitat, and associated references.

Dead Wood Density (per 0.04 ha)	SI Score (EUST)	Reference
0	0.06	Strauss et al., 2011
1	0.93	Strauss et al., 2011
3	1	Adkins et al., 2003

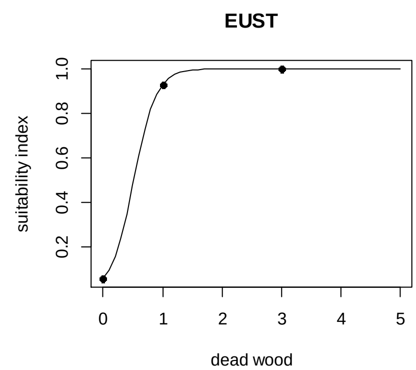


Table 12. Relationship between canopy percent per 0.04 ha and suitability index (SI) for northern cardinal (NOCA) habitat, and associated references.

Canopy Percent (per 0.04 ha)	SI Score (NOCA)	Reference
0	0.7	assumed value
10	0.8	assumed value
20	0.9	assumed value
27	1	Conner et al., 1986
30	1	assumed value
32	1	assumed value
40	0.6	assumed value
50	0.4	assumed value
70	0.2	assumed value
90	0.1	assumed value
100	0	assumed value

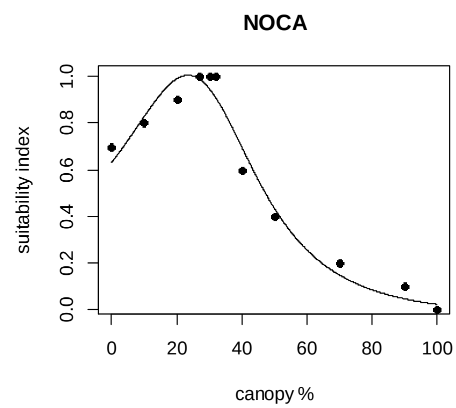


Table 13. Relationship between shrub percent and suitability index (SI) for northern cardinal (NOCA) habitat, and associated references.

Shrub Percent (per 0.04 ha)	SI Score (NOCA)	Reference
0	0	assumed values
20	0.6	assumed values
40	0.8	assumed values
60	1	assumed values
80	1	assumed values
100	0.8	assumed values

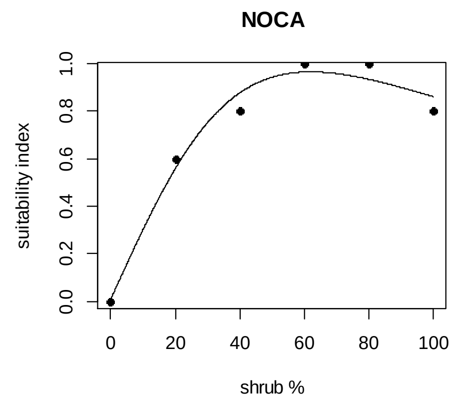


Table 14. Relationship between large tree density (trees larger than 23 cm dbh) per 0.04 ha and suitability index (SI) for red-bellied woodpecker (RBWO) habitat, and associated references.

Large Tree density (per 0.04 ha)	SI score (RBWO)	Reference
0	0	Assumed value
3	0.6	Assumed value
6	1	Adkins et al., 2003
8	0.9	Conner, 1980
11	0.8	Assumed value

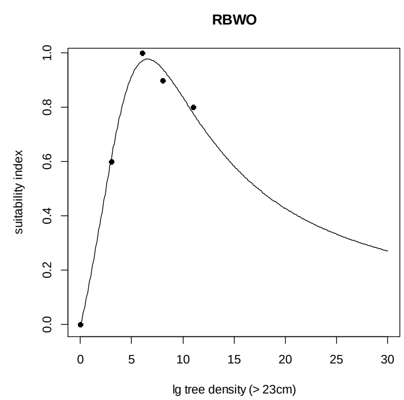


Table 15. Relationship between basal area (trees > 6 cm dbh) per ha and suitability index (SI) for red-bellied woodpecker (RBWO) habitat, and associated references.

Basal Area (per ha)	SI Score (RBWO)	Reference
0	0	Assumed value
4	0.5	Assumed value
8	0.95	Conner, 1980 (based on SD)
14	1	Conner, 1980
34	1	Adkins et al., 2003

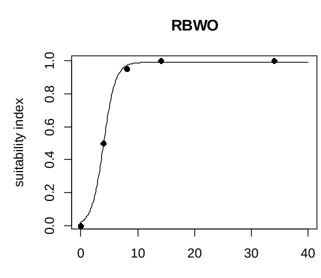


Table 16. Relationship between canopy percent per 0.04 ha and suitability index (SI) for red-bellied woodpecker (RBWO) habitat, and associated references.

Canopy Percent (per 0.04 ha)	**SI Score (RBWO)**	**Reference**
0	0	assumed value
15	0.1	assumed value
20	0.3	assumed value
25	0.5	assumed value
35	0.9	DeGraaf et al., 2006
62	1	Strauss et al., 2011

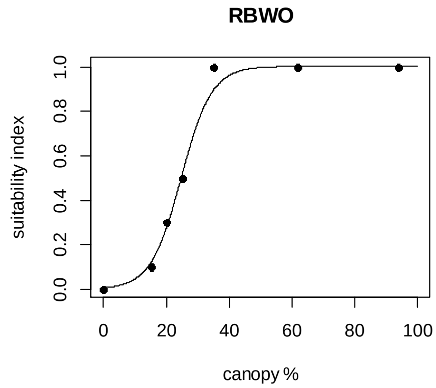


Table 17. Relationship between dead wood density per ha and suitability index (SI) for red-bellied woodpecker (RBWO) habitat, and associated references.

Dead Wood Density (per 0.04 ha)	**SI Score (RBWO)**	**Reference**
0	0.06	Strauss et al., 2011
1	0.93	Strauss et al., 2011
3	1	Adkins et al., 2003

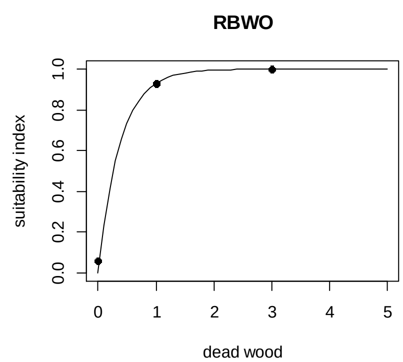


Table 18. Relationship between large tree density (trees larger than 23 cm dbh per 0.04 ha) and suitability index (SI) for scarlet tanager (SCTA) habitat, and associated references.

Large Tree Density (per 0.04 ha)	SI Score (SCTA)	Reference
0	0	assumed value
10	0.1	assumed value
15	0.4	assumed value
20	1	Roberts & Norment, 1999
26.2	1	Shy, 1984
32	1	Rivera et al., 2003

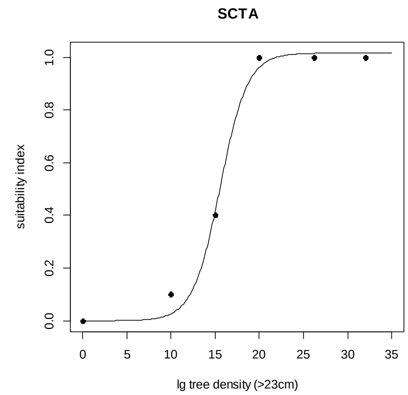


Table 19. Relationship between basal area (trees >6 cm dbh) per ha and suitability index (SI) for scarlet tanager (SCTA) habitat, and associated references.

Basal Area (per ha)	SI Score (SCTA)	Reference
0	0	Assumed value
10	0.05	Assumed value
20	0.2	Assumed value
30	0.3	Assumed value
40	0.7	Assumed value
50	0.8	Assumed value
62	1	Roberts & Norment, 1999

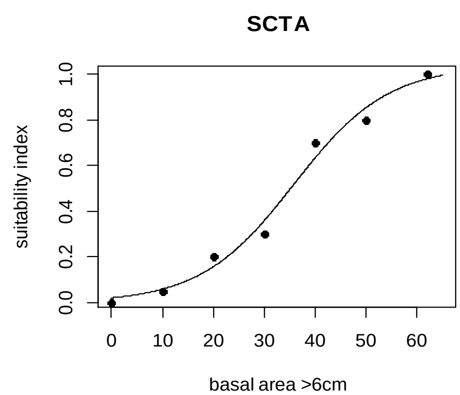


Table 20. Relationship between canopy percent per 0.04 ha and suitability index (SI) for scarlet tanager (SCTA) habitat, and associated references.

Canopy Percent (per 0.04 ha)	SI Score (SCTA)	Reference
0	0	assumed value
50	0.2	assumed value
75	0.95	Shy, 1984
89	1	Shy 1984, Roberts and Norment, 1999

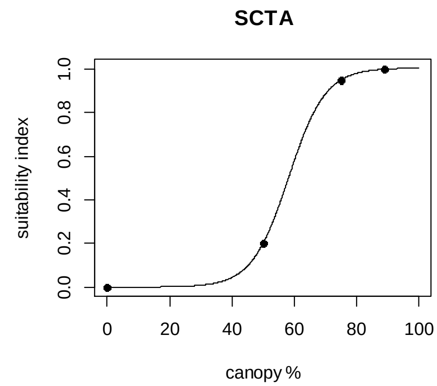


Table 21. Relationship between contiguous forest area (ha) surrounding the i-Tree plot and suitability index (SI) for scarlet tanager (SCTA) habitat, and associated references.

Forest Area (ha)	SI Score (SCTA)	Reference
0	0	assumed value
1	0.3	Robbins et al., 1989
10	0.5	Robbins et al., 1989
32	0.6	Robbins et al., 1989
100	0.7	Robbins et al., 1989
300	0.8	Robbins et al., 1989
500	0.9	Robbins et al., 1989

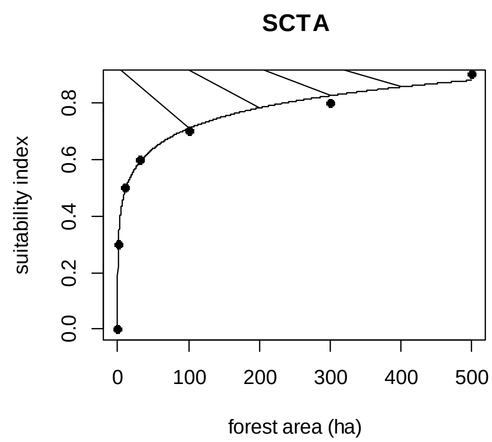


Table 22. Relationship between canopy percent per 0.04 ha and suitability index (SI) for wood thrush (WOTH) habitat, and associated references.

Canopy Percent (per 0.04 ha)	SI Score (WOTH)	Reference
25	0	Hoover & Brittingham, 1998
70	0.25	Annand & Thompson, 1997
90	0.9	Annand & Thompson, 1997
100	1	Annand & Thompson, 1997

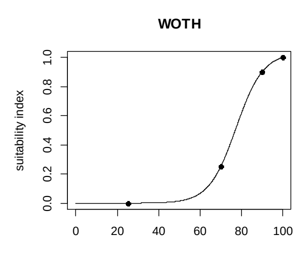


Table 23. Relationship between sapling density and suitability index (SI) for wood thrush (WOTH) habitat, and associated references.

Sapling Density (per 0.04 ha)	SI Score (WOTH)	Reference
0	0	assumed value
2	0.1	assumed value
5	0.4	assumed value
7	0.8	assumed value
10	1	assumed value

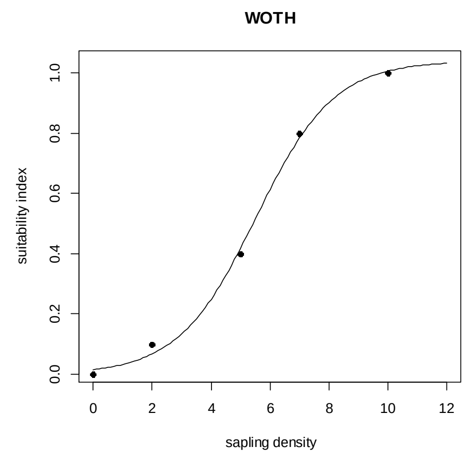
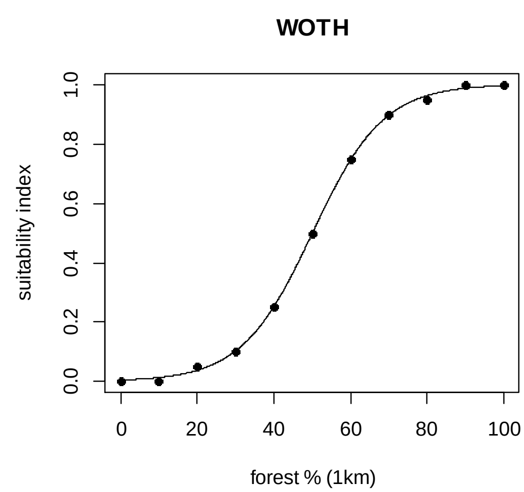


Table 24. Relationship between percent forest within a1km radius of the i-Tree plot and suitability index (SI) for wood thrush (WOTH) habitat, and associated references.

% Forest (1km)	SI Score (WOTH)	Reference
0	0	assumed value
10	0	assumed value
20	0.05	assumed value
30	0.1	Donovan et al., 1997
40	0.25	assumed value
50	0.5	Donovan, et al., 1997
60	0.75	assumed value
70	0.9	Donovan et al., 1997
80	0.95	assumed value
90	1	Donovan et al., 1997
100	1	assumed value



Supplementary Material
Species Accounts and Models

American Robin

Status

The American Robin *Turdus migratorius* is one of North America's most abundant, widespread and recognizable birds. This familiar migratory species thrives in both suburban and wildland settings, and has been deemed "America's favorite songbird" (Sharp, 1990). American Robin populations are either stable or increasing throughout the range; deforestation, urbanization and agricultural development often create habitat for the species (Sallabanks & James, 1999). Due to its ability to thrive in a variety of habitat conditions, the American Robin does not have a conservation status. However, it is used as a bioindicator for chemical pollution. The species has enjoyed a 1.9 percent increase between 1966-1996 (Sallabanks & James, 1999).

Natural History

A lower canopy, shrub forager, the American robin's diet varies seasonally between earthworms and other soft invertebrates in spring and summer, and fruit in fall and winter (Sallabanks & James, 1999). Foraging substrates include lawns, loamy soil, and fruit bearing trees, shrubs and vines. Breeding habitat ranges from open woodlands and woodland edges and clearings, fields, orchards, and shade trees in residential areas. Residential areas and parks with lawns interspersed with shrubs and trees are ideal. Nesting sites vary and include horizontal branches or forks of a tree, shrubs and ledges of buildings. The sky blue eggs and speckled nestlings are familiar to many suburbanites. Territory sizes vary with population density, ranging between 0.04 and 0.84 ha (Pitts, 1984; Young, 1950). Winter territorial behavior focuses around the defense of fruit (Young, 1950).

Model Description

The habitat suitability index (HSI) model for the American robin includes two plot variables: percent canopy cover and percent lawn cover as estimated for a 0.04 ha plot. Although forest area has been shown to limit robin populations (Keller, Robbins, & Hatfield, 1993; Robbins, Dawson, & Dowell, 1989), we believe that local features are better predictors for this species.

We based our assumed values for canopy percent on qualitative accounts of the species requiring some trees yet not requiring extensive woodlands (Table 1). We fit a rational function (0.6439+(-0.0024*Canopy Percent))/(1+(-0.0312*Canopy Percent)+(0.0005*Canopy Percent^2)) through these data points to quantify the relationship between canopy coverage and the suitability index (SI score; Fig. 1).

We based our assumed values for lawn percent on qualitative accounts and personal observations of the species extracting earthworms and other soft invertebrates from manicured lawns in wooded parks and residential yards (Table 2). Since the robin primarily nests in trees and shrubs, an area with 100 % lawn cover would not be suitable. The relationship reflects the inverse of percent canopy. We fit a reciprocal quadratic function (1/(4.1918+(-0.0831*Lawn

Percent)+(0.00051* Lawn Percent ^2)) through these data points to quantify the relationship between lawn coverage and the suitability index (SI score; Fig. 2). We calculated the geometric mean of these habitat models to generate a final SI score for this species.

Baltimore Oriole

Status

The Baltimore oriole (*Icterus galbula*) is a long-distance neotropical migrant found throughout north eastern and central United States, and the plains of Canada. This species has adapted well to suburbia and urban parks, and thus, is another of America's most familiar songsters. The management of treed parks in urban and suburban areas will assist with the broadening of the breeding distribution (Ickes, 1992). The species has a Partners in Flight (PIF) score of 17 (a score of 30 being the highest for this region and thus having the highest level of conservation concern) in the mid-Atlantic region and a 3.2 percent decline between 1966-1996 (Watts, 1999). The PIF score in southern New England is 23 (Dettmers & Rosenberg, 2000).

Natural History

This canopy-gleaning passerine is found in a variety of habitats, favoring deciduous woodland edges, especially along riparian corridors, and suburban areas with tall and scattered shade trees, groves, orchards and parks. Also found in open woodlands with well-spaced trees (Salt & Salt, 1976); avoids closed-canopy forests (Palmer-Ball, 1996), and prefers large trees (e.g. dbh > 23 cm; Perkins, Johnson, & Blankenship, 2003). The species nests in deciduous trees, and builds their pendulant nest in the upper canopy, near the tip or outer branch of a tree (Rising & Flood, 1988). The familiar pendulant nest droops down from the upper branches. Territory sizes range from 0.15 ha to 1.86 ha.

Model Description

The HSI for Baltimore oriole includes one plot variable: canopy percent within a 0.04 ha plot and one tree variable: tree density for trees greater than 23 cm diameter at breast height. These variables address the relationship between a moderate canopy cover that consists of primarily large, open-grown trees.

We based our assumed values for canopy percent on qualitative accounts of the species requiring some trees yet not requiring extensive woodlands (Table 3). We fit a gausian function 1.0127*exp(0-((Canopy Percent-35.4635)^2)/(2*15.3508^2)) through these data points to quantify the relationship between canopy coverage and the suitability index (SI score; Fig. 3).

We based our assumed values for the upper limits of large tree density (>23 cm dbh) per 0.04 ha plot on i-Tree data sets: four of the ten cities had plots with at least 11 large trees present (Table 4). In addition, a higher density of large trees would increase the canopy coverage for the plot. The tree size also positively correlates with tree height and therefore larger trees are taller and thus more suitable for this high-canopy nester. We fit a rational function (0.0378+(0.2794*large tree density))/(1+(-0.4471* large tree density)+(0.1311* large tree density ^2))through these data points to quantify the relationship between large tree density and the suitability index (SI score; Fig. 4). We calculated the geometric mean of these habitat models to generate a final SI score for this species.

Black-capped Chickadee
Carolina Chickadee

Status

The black-capped chickadee (*Parus atricapillus*) is one of America's most widespread and familiar species. This non-migratory species can be found throughout the northern half of the United States and much of Canada. Based on Breeding Bird Surveys, eastern populations are thought to be increasing, though the range expansion of tufted titmouse (*Baeolophus bicolor*) might negatively impact chickadee populations (Loery & Nichols, 1985; Smith, 1991). Although urbanization has a negative effect on black-capped chickadees, suburban areas with large natural snags seem to partially mitigate the impacts of urban development (Blewett & Marzluff, 2005; Donnelly & Marzluff, 2006). The species has a PIF score of 15 in the mid-Atlantic region (Watts, 1999). No PIF scores reported for southern New England (Dettmers & Rosenberg, 2000).

The Carolina chickadee (*Parus carolinensis*) is the southeastern counterpart of black-cappped chickadee, with western limits in Kansas and eastern Texas and northern reaches into New Jersey and Pennsylvania. Similar to black-capped chickadees, the Carolina chickadee has adapted to suburbanization due to the presence of bird feeders and nest boxes (Doherty & Grubb 2002a; Hadidian, Sauer, Swarth, Handly, Droege, Williams, Huff, & Didden, 1997; Ringler, 1996). However, suburban areas highly prone to habitat fragmentation and areas with strong, negative interactions with house wrens (*Troglodytes aedon*) might lead to negative population trends (Doherty & Grubb, 2002b; Foote, Mennill, Ratcliffe, & Smith, 2010; Mostrom, Curry, & Lohr, 2002). Carolina chickadee has a PIF score of 21 and a 2.2 percent decline between 1966 and 1996 in the mid Atlantic region (Watts, 1999).

Natural History

A lower canopy, shrub gleaner, both the black-capped and Carolina chickadee diet consists mainly of insects during the breeding season and a mixture of seeds and berries, and insects and spiders during the winter (Smith, 1991; 1993). Breeding habitat includes deciduous, coniferous, or mixed woodlands (mixed preferred for black-capped, deciduous preferred for Carolina; Morse, 1970), and both species can be found in heavily forested and residential areas, with optimal conditions of an open understory and mature subcanopy (Anderson & Shugart, 1974). Wintering habitat includes city parks and residential areas with feeding stations adjacent to breeding habitat. Specific habitat requirements include dead standing trees or stubs (minimum dbh 10 cm; Holmes, 2002) for excavating cavities or trees with existing cavities for nesting (Mostrom, Curry, & Lohr, 2002). The chickadees will also use nest boxes.

Model Description

The US Fish and Wildlife Service developed an HSI model for black-capped chickadee which included percent canopy cover (i-Tree plot variable), tree height and dead wood density (i-Tree

tree variable; Holmes, 2002; Schroeder, 1982). We used the revised models published by Holmes (2002) for the i-Tree datasets. Due to the similarities in habitat requirement (Mostrom, Curry, & Lohr, 2002), we assumed that the variables thought to limit population abundances of black-capped chickadee was applicable to Carolina Chickadee and therefore present one model for both species.

Sturman (1968) proposed that tree foliage volume was a strong predictor of arthropod abundance, a major food resource for chickadees. However, this is time consuming to measure, and therefore Schroeder (1983) suggested that canopy cover and mean tree height per 0.04 ha plot were suitable alternative variables to address food resources (Table 5, 6). Although these data are available in i-Tree, we stuck with Schroeder's models. We fit a gausian function (1.002*exp((0-((Canopy Percent)-63.5682)^2)/1795) through these data points to quantify the relationship between canopy coverage and the suitability index (SI score; Fig. 5). We fit a logistic function (0.9757/(1+(11.7426*exp(-0.4852*Mean Tree Ht(m)))) through the data points to quantify the relationship between mean tree height (m) per 0.04 ha plot and the suitability index (SI score; Fig. 6).

To address nesting resources we calculated the number of trees within a 0.04 ha plot with deadwood. To calculate the density of dead wood per i-Tree plot, we assumed all trees with a condition classification of "fair", "poor", "dying" and "dead" potentially harbored wood that could be excavated for cavities. In addition to Holmes' (2002) values, we included Sedgwick and Knopf (1990) data that sites with at least six trees with dead wood were most suitable (Table 7). We fit a logistic function (1.007/(1+(32.567*exp(-1.403*density of trees with dead wood))) through the data points to quantify the relationship between density of trees with deadwood per 0.04 ha plot and the suitability index (SI score; Fig. 7). We calculated the geometric mean of these habitat models to generate a final SI score for this species.

European Starling

Status

From humble beginnings of about 100 individuals released in Central Park, NY, the European Starling (*Sturnus vulgaris*) is one of the most successful introductions to North America with a population hovering around 200 million (Cabe, 1993). Although declining in Europe (Newton, 2004), this species continues to thrive throughout North American cities, towns and agricultural areas. Due to its overabundance, this species is considered a nuisance and efforts to control populations are common, though often ineffective. Starlings often compete with native species for cavity sites and thus their increasing presence has detrimental impacts on other cavity-nesting species (Cabe, 1993). The species has a PIF score of 11 and a 2.7 decline between 1966-1996 in the mid-Atlantic region (Watts, 1999). No conservation status provided for southern New England (Dettmers & Rosenberg, 2000).

Natural History

A ground-foraging omnivorous passerine, the European starling is one of North America's most synanthropic species. Urban (e.g. lawns) and cultivated fields and hayfields, orchards, and parks provide ideal conditions for this species and can often be seen foraging for insects, grains, fruit and seeds in these areas. They appear to avoid pristine wilderness areas including non-grasslands, forests, deserts and arid chaparral (Cabe, 1993). They form massive winter roosts in dense vegetation, with numbers exceeding a million individuals. The shimmering flight pattern of a tight flock is a common spectacle over fields and cities alike. This cavity nester will use a variety of holes including crevices in buildings, cliffs, nest boxes and previously occupied woodpecker cavities. The territories are focused within the immediate vicinity of the cavity entrance (ca. 50 cm; Kessel, 1957).

Model Description

The habitat suitability index (HSI) model for the European starling includes two plot variables: percent lawn cover and percent building cover as estimated for a 0.04 ha plot, and two tree variables: tree density and density of deadwood present within a 0.04 ha plot. Although forest area has been shown to limit starling populations (Keller, Robbins, & Hatfield, 1993; Robbins, Dawson, & Dowell, 1989), we believe that local features are better predictors for this species.

We based our assumed values for lawn percent on qualitative accounts and personal observations of the species extracting insects and seed from manicured lawns in wooded parks and residential yards to areas with extensive amounts of lawn and turf (Table 8). We fit a logistic function (1.02247436719/(1+(40.643183849*exp(-0.1043766533*lawn percent per 0.04 ha))) through these data points to quantify the relationship between lawn coverage and the suitability index (SI score; Fig. 8).

We based our assumed values for building percent on qualitative accounts and personal observations of this species nesting in rain gutters, eaves, and other building cavities. Since the starling is extremely synanthropic, we assumed that plots with 100% building cover were suitable despite the presence of lawn (Table 9). We fit a rational function (-0.0004+(0.0148*building percent per 0.04 ha))/(1+(-0.0379* building percent per 0.04 ha)+(0.0007* building percent per 0.04 ha ^2)) through these data points to quantify the relationship between building coverage and the suitability index (SI score; Fig. 9).

We based our assumed values for tree density per 0.04 ha to reflect a gradient from field to forest, with areas of low tree density (all size classes) being more suitable (Table 10). In addition, tree density reflects the inverse relationship with lawn percent. We fit a rational function (0.8129+(-0.088*tree density per 0.04 ha))/(1+(-0.3167* tree density per 0.04 ha)+(0.0547* tree density per 0.04 ha ^2)) through these data points to quantify the relationship between tree density and the suitability index (SI score; Fig. 10).

European starling also nests in cavities and we assumed some conditions suitable for woodpeckers (e.g. red-bellied woodpecker) would also apply for starlings. Straus, Bavrlic, Nol, Burke, and Elliott, (2011) found that when at least one tree with deadwood was present within a 0.04 ha plot, red-bellied woodpeckers were also present. Adkins, Giese, and Cuthbert (2003) found slightly higher densities of deadwood to be more suitable. Due to starlings' ability to nest in artificial cavities (i.e. not tree cavities), we assumed that sites without deadwood present were still suitable (Table 11). We fit an exponential function (0.8005*(1.2498-exp(-2.4290*density of trees with deadwood)) through these data points to quantify the relationship between deadwood density within 0.04 ha and the suitability index (SI score; Fig. 11). We then calculated the geometric mean of these habitat models to generate a final SI score for this species.

Northern Cardinal *Cardinalis cardinalis*

Status

The Northern Cardinal (*Cardinalis cardinalis*) is resident found throughout eastern and central United States and Mexico. Northward movements of its range have been attributed to the urban heat island effect and provisions at bird feeders. The brilliantly red male is one of America's most familiar birds and is the state bird for seven states (Halkin & Linville, 1999). Alteration of habitat (converting forests to agriculture and suburbs) has benefited cardinals by increasing nesting habitat (Halkin & Linville, 1999). Based on Breeding Bird Survey data, the species has experienced 1 percent decline between 1966-1996 and has a PIF score of 14 in the Mid-Atlantic region. The species has experienced a 3.3 percent increase between 1966 and 1996 (Watts, 1999), and no PIF score for southern New England (Dettmers & Rosenberg, 2000).

Natural History

This omnivorous ground gleaner's diet consists of seeds, fruit, and insects (Halkin & Linville, 1999). Bird feeding during winter assists with survival in the colder parts of its range. Cardinals can be found along forest edges, open woodlands, suburban yards, urban parks and other areas with thickets and shrubs. In an extensive study of a breeding bird community in east Tennessee, Anderson and Shugart (1974) found that cardinals preferred sites with a thick subcanopy and relatively open canopy. Nest sites are located in dense, low vegetation including shrubs and small trees (deciduous and coniferous), vines, thickets and briars (Conner, Anderson, & Dickson, 1986; Ehrhart & Conner, 1986), and with prominent song posts in close proximity (Dow, 1969). Territory size ranges from 0.21 to 2.60 ha (Halkin & Linville, 1999).

Model Description

The habitat suitability model (HSI) for northern cardinal includes two plot variables: canopy percent and shrub percent, both within a 0.04 ha plot. These variables address the relationship between a moderate to open canopy cover which can encourage an extensive shrub layer.

We based our assumed values for canopy cover on qualitative accounts and personal observations of the species in edge habitats, residential yards with little to no canopy cover and lack of observations in thick, extensive woodlands (Table 12). We fit a rational function (0.6313+(-0.0054* Canopy Percent))/(1+(-0.0370* Canopy Percent)+(0.0007* Canopy Percent ^2)) through the data points to quantify the relationship between percent canopy cover per 0.04 ha plot and the suitability index (SI score; Fig. 12).

We based our assumed values for shrub cover on qualitative accounts and personal observations of the species nesting in dense shrubs, privets, thickets and other low vegetation in residential yards and urban parks (Table 13). We fit a rational function (0.0095+(0.0213*Shrub Percent))/(1+(-0.0212*Shrub Percent)+(0.0004*Shrub Percent ^2)) through the data points to quantify the relationship between percent shrub cover per 0.04 ha plot and the suitability index (SI score; Fig. 13).

Red-bellied Woodpecker

Status

The red-bellied woodpecker (*Melanerpes carolinus*) has a broad distribution throughout the eastern half of the United States. The species is resident throughout the eastern part of its range though northern birds move south during cold winters (Winkler, Christie, & Nurney, 1995). The red-bellied woodpecker is a familiar site to feeder watchers and easily recognized. The species is commonly reported throughout the eleven i-Tree cities and not of conservation concern largely in part to its preference for a wide range of forest types (Shackelford, Brown, & Conner, 2000). Based on Breeding Bird Survey data, the nationwide population is either stable or increasing (Price, Droege, Price, & Beadle, 1995) and appears to thrive in urban and suburban areas. However, similar to other woodpeckers, the red-bellied is heavily dependent on snags and dead wood for nesting and roosting and therefore urban forest management plans that encourage dead wood have the potential to support this species. For the mid-Atlantic region, the species has a PIF score of 15 (Watts, 1999), and in southern New England, the species has increased 21.1 percent between 1966 and 1996 (Dettmers & Rosenberg, 2000).

Natural History

The red-bellied woodpecker is a vocal and conspicuous cavity-nester found in mature pine forests, hardwood forests or a mixture of the two. The species excavates cavities in snags on dead trees or dead limbs on live trees. An opportunistic forager, this species' diet consists of fruit, beech and acorn masts and arboreal arthropods. Red-bellied woodpeckers are sedentary, remaining on breeding grounds year-round. Average territory size ranges from 1.8 to 2.5 ha based on studies from upland forests and virgin floodplain forest in Illinois (Shackelford, Brown, & Conner, 2000).

Model Description

The habitat suitability index model for the red-bellied woodpecker includes four plot variables: tree density per 0.04 ha, basal area per ha, density of dead wood (i.e. trees classified as fair, poor, dying or dead) per 0.04 ha, and percent canopy cover per 0.04 ha.

The species relies on forested areas and we included three variables to describe these habitat needs. Adkins, Giese, and Cuthbert (2003) observed 24 trees per 0.04 ha and a basal area of 34 m^2 per ha in oak forests of the Upper Midwest, while Conner (1980) observed 30 trees per 0.04 ha and a basal area of 14 m^2 per ha in oak-hickory forests around Blacksburg, VA. However, these studies didn't discern tree size. We wanted the model to reflect the mean diameter of the cavity limb (21.6 cm; Jackson, 1976) so only included trees greater than 23 cm dbh and adjusted the densities to reflect these conditions (Table 14). We fit a rational function (0-0.0035+(0.1606*Tree Density))/(1+(-0.1417*Tree Density)+(0.0233* Tree Density ^2)) where

Tree Density is the density of trees greater than 23 cm dbh within a 0.04 ha plot, through these data points to predict how habitat suitability varied with large tree density (Fig. 14). We assumed suitability was the lowest when trees were absent. Our inclusion of basal area for all trees greater than 6 cm dbh reflects the propensity for this species to prefer relatively dense forests (Shackelford, Brown, & Conner, 2000; Table 15). We fit a logistic function 0.9906/(1+(47.9216*exp(-0.9689*basal area))) where basal area is m^2/ha and calculated for all trees greater than 6 cm dbh, through these data points to quantify the relationship between basal area and the suitability index (SI score; Fig. 15).

Canopy coverage also predicts habitat suitability. DeGraaf, Yamasaki, Leak, and Lester, (2006) suggested that when canopy coverage exceeding 35%, the site provided suitable conditions for red-bellied woodpeckers. We based our assumed values for canopy cover on qualitative accounts and personal observations of the species in forested suburban and riparian areas, with lack of observations in areas with little to no canopy cover and areas with an extremely dense canopy cover (Table 16). We fit a rational function (-0.0371+(0.0124*Percent Canopy))/(1+(-0.0363* Percent Canopy)+(0.0005* Percent Canopy ^2)), where Percent Canopy is the percent of a 0.04 ha plot with tree canopy cover, through these data points to predict how habitat suitability varied with canopy coverage (Fig. 16). We assumed suitability was the lowest when trees were absent.

Although dead wood is necessary for foraging and nesting, they are not essential for detecting red-bellied woodpeckers. Of 42 nests in southwest Ontario, Strauss et al. (2011) observed 93% of the nests in dead and declining trees and 6% of nests in healthy trees. Adkins, Giesse, and Cuthbert (2003) observed 3 dead or declining trees per 0.04 ha in the Midwest (Table 17). We fit a logistic function 1/(1+(15.67*exp(-5.338*Dead Wood density per 0.04 ha))), (where dead wood is recorded as trees with a condition of fair, poor, dying or dead) through these data points to quantify the relationship between trees with dead wood and the suitability index (SI score; Fig. 17). We calculated the geometric mean of these habitat models to generate a final SI score for this species.

Scarlet Tanager

Status

The scarlet tanager (Piranga olivacea) is a long-distance neotropical migrant, found in deciduous forests throughout the northeastern United States and southern Canada. This forest interior species is highly sensitive to forest fragmentation (Roberts & Norment, 1999). In a study from New Jersey, scarlet tanagers were present only in forest fragments greater than 3 ha, though forest areas greater than 10 ha were required to sustain a viable population (Galli, Leck, & Forman, 1976; Robbins, Dawson, & Dowell, 1989; Roberts & Norment, 1999). As fragment size decreases, nest predation and parasitism rates increase (Robinson, Thompson III, Donovan, Whitehead, & Faaborg, 1995). According to the Breeding Bird Survey, the species has experienced a1.6 percent decline between 1966 and1996, and has a PIF score of 21 in the mid-Atlantic region (Watts, 1999). The species has experienced a 1 percent decline and has a PIF score of 22 in southern New England (Dettmers & Rosenberg, 2000).

Natural History
Scarlet tanagers spend most of their time in the mid to upper canopy, hovering and gleaning insects from flowers, fruit, leaves and bark (Mowbray, 1999). They are associated with mature deciduous and mixed forests but occasionally found in dense shade trees in suburban areas, cemeteries and parks (Mowbray, 1999). They prefer trees greater than 22.4 cm dbh, and primarily in oak-hickory woods (Mowbray, 1999). Territory size varies according to vegetation type but ranges from 0.8 – 5.0 ha (Robbins, 1980; Zumeta & Holmes, 1978).

Model Description
The HSI model for the scarlet tanager includes three plot variables: large tree (> 23 cm dbh) density per 0.04 ha, basal area per ha, and percent canopy cover per 0.04 ha. The model also includes one landscape variable: extent of forest patches (ha).

Three separate studies found that when a forested 0.04 ha plot had at least 20 large trees, the area would support scarlet tanagers (Shy, 1984; Roberts & Norment, 1999; Rivera, McShea, & Rappole, 2003; Table 18). We fit a logistic function (1.01622702/(1+(24569.22035*EXP(-0.6493929*Tree Density))) where Tree Density is the density of trees greater than 23 cm dbh within a 0.04 ha plot, through these data points to predict how habitat suitability varied with large tree density (Fig. 18). We assumed suitability was the lowest in plots with fewer than ten trees. In addition to mature forests (i.e. those with large trees), scarlet tanagers also prefer dense forests. Based on Roberts and Norment (1999) we calculated a mean basal area of 62 m^2/ha and assumed this density to be most suitable for tanagers (Table 19). We fit a logistic function (1.0363/(1+(49.295*EXP(-0.1088*Basal Area))) where basal area is m^2/ha and calculated for all trees greater than 6 cm dbh, through these data points to quantify the relationship between basal area and the suitability index (SI score; Fig. 19).

Scarlet tanagers prefer territories in forested areas with dense canopy cover (Ambuel & Temple, 1983). Roberts and Norment (1999) and Shy (1984) suggest that a canopy coverage of 89% represented the most suitable conditions for this species, with forested patches having 75% canopy coverage highly suitable (Shy, 1984; Table 20). We fit a logistic function (1.0363/(1+(49.295*EXP(-0.1088*Percent Canopy))) where Percent Canopy is the percent of a 0.04 ha plot with tree canopy cover, through these data points to predict how habitat suitability varied with canopy coverage (Fig. 20). We assumed suitability declined significantly when canopy coverage was less than 50%.

We included one landscape variable to account for the extreme sensitivity this species exhibits to forest fragmentation (Robinson et al., 1995; Table 21). We fit a multiple multiplicative factor (MMF) function ((-0.0009*4.3992)+(1.6780*Forest Area ^0.2539))/(4.3992+ Forest Area ^0.2539) where Forest Area is forest patch size (ha) through these data points to predict how habitat suitability varied with the extent of forested areas (Fig. 21). Although a low suitability, forest patches 1 ha have the potential to harbor scarlet tanagers, thus suggesting the possibility of urban remnant patches to have some conservation value for this species. We calculated the geometric mean of these habitat models to generate a final SI score for this species.

Wood Thrush

Status

The wood thrush (Hylocichla mustelina) is a long-distance neotropical migrant found throughout eastern North America (Roth, Johnson, & Underwood, 1996). This species is a symbol of threatened and declining songbirds and it has become increasingly rare throughout its range since the 1970s (Evans, Gow, Roth, Johnson, & Underwood, 2011). Habitat loss and fragmentation in the breeding and wintering grounds have had detrimental effects on populations, and thus exacerbating the impacts of cowbird parasitism (Thompson III, Robinson, Donovan, Faaborg, Whitehead, & Larsen, 2000). The wood thrush is a species of conservation concern, with a 2.3 percent decline between 1966 and 1996, and a Partners in Flight score of 24 for the mid Atlantic (Watts, 1999). The species has experienced a 2.2 percent decline between 1966 and 1996, and PIF score of 24 for southern New England (Dettmers & Rosenberg, 2000).

Natural History

A ground-foraging passerine, the wood thrush is associated with mature upland forests (mainly deciduous or mixed but largely avoids evergreen stands) with closed overstory canopies (Bell & Whitmore, 2000; Evans et al., 2011). Additional conditions include forest patches with trees taller than >16 m, moderate subcanopy, sapling density and shrubs for nesting, cool and moist soil conditions, and a somewhat open forest floor with decaying leaf litter for foraging (Evans et al., 2011). The species is thought to be highly sensitive to forest fragmentation with regards to its productivity but has nested in small forest fragments (0.3 ha; e.g. remnant patches in residential areas and parks) at low densities (Tilghman, 1987; Weinberg & Roth, 1998).

The wood thrush's diet mainly consists of soil invertebrates and fruits, and occasionally feeds on arboreal insects, snails and small salamanders (Evans et al., 2011). The nest is typically located on a horizontal branch or crotch within a sapling or tree (Evans et al., 2011). Hoover and Brittingham (1998) suggested that nest success was better predicted by the amount of forest in the landscape rather than the microhabitat structural features surrounds nests. Territory size ranges between 0.08 and 4.0 ha (Evans, Stutchbury, & Woolfenden, 2008; Twomey, 1945).

Habitat Model

The HSI model for the wood thrush includes two plot variables: percent canopy cover per 0.04 ha and sapling density per 0.04 ha plot. The model also includes one landscape variable: percent forest landcover within a 1 km radius.

The wood thrush associates with dense canopied forests (Table 22) and we fit a logistic function 1.03163/(1+(141241.64*EXP(-0.1531*Percent Canopy))) where Percent is the percent of a 0.04 ha plot with tree canopy cover to data from Annand and Thompson (1997) and Hoover and Brittingham (1998) to predict SI scores from percent canopy coverage scores (Fig. 22). Tirpak et al. (2009) devised a model that incorporated small stem densities (< 2.5 cm) based on Hoover and Brittingham (1998) assertion that 1,988 stems per ha represented optimal habitat. These stem densities within the i-Tree datasets were far from this abundance, even when we included shrub densities into the equation. We therefore used sapling density (<10 cm) as a proxy to assess the midstory cover. We based our assumed values on existing i-Tree datasets and balanced the relationship between a dense canopy cover and the sapling density whereby at least 10 saplings were recorded in i-Tree plots with 90% canopy coverage (Table 23). We fit a logistic function (1.0401978/(1+(65.800186*EXP(-0.758149*(Sapling Density))))), where sapling density is the number of trees <10 cm in diameter within a 0.04 ha plot, through these data points to quantify the relationship between sapling density and the suitability index (Fig. 23).

Although studies have demonstrated the importance of forest area as a predictor of nest success for wood thrush (e.g. Robbins, Dawson, & Dowell, 1989; Kilgo et al., 1998), we chose a variable that reflected the amount of forest patches within the greater landscape matrix to address the suitability of urban parks and remnant patches within residential landscapes (Table 24). Following Tirpak et al. (2008), we fit a logistic function 1.003/(1+(224.7853*EXP(-0.1081*(1KM % Forest)))) where 1KM % Forest is the percent of a 1 km plot around an i-Tree monitoring plot classified as forest landcover to data based on Donovan, Jones, Annand, and Thompson (1997; Fig. 24). In this study, the predator and brood parasite communities were related to fragmentation size: highly fragmented (< 15 percent), moderately fragmented (45 to 50 percent), and lightly fragmented (> 90 percent forest) landscapes. We followed logic from Tirpak et al. (2008), and also assumed the midpoints between 30 and 70 percent represented the thresholds for low suitability (SI score ≤ 0.10) and excellent suitability (SI score ≥ 0.90) habitats. We calculated the geometric mean of these habitat models to generate a final SI score for this species.

Literature Cited

Adkins Giese, C. L., & Cuthbert, F. J. (2005). Woodpecker nest tree characteristics in upper midwestern oak forests. *The Canadian Field-Naturalist*, 119(3), 367–376.

Ambuel, B., & Temple, S. A. (1983). Area-Dependent Changes in the Bird Communities and Vegetation of Southern Wisconsin Forests. *Ecology*, 64(5), 1057–1068. doi:10.2307/1937814

Anderson, S., & Shugart, H. (1974). Habitat selection of breeding birds in an east Tennessee deciduous forest. *Ecology*, 55(4), 828–837. doi:10.2307/1934418

Annand, E. M., & Thompson, F. R. (1997). Forest Bird Response to Regeneration Practices in Central Hardwood Forests. *The Journal of Wildlife Management*, 61(1), 159–171. doi:10.2307/3802425

Bell, J. L., & Whitmore, R. C. (2000). Bird Nesting Ecology in a Forest Defoliated by Gypsy Moths. *The Wilson Bulletin*, 112(4), 524–531.

Blewett, C. M., & Marzluff, J. M. (2005). Effects of Urban Sprawl on Snags and the Abundance and Productivity of Cavity-Nesting Birds. *The Condor*, 107(3), 678–693.

Cabe, P. R. (1993). European Starling (Sturnus vulgaris). (A. Poole, Ed.). *The Birds of North America Online*. Retrieved from http://bna.birds.cornell.edu/bna/species/048 doi:10.2173/bna.48

Conner, R. (1980). Foraging habits of woodpeckers in southwestern Virginia. *Journal of Field Ornithology*, 51(2), 119–127.

Conner, R. N., Anderson, M. E., & Dickson, J. G. (1986). Relationships among Territory Size, Habitat, Song, and Nesting Success of Northern Cardinals. *The Auk*, 103(1), 23–31.

DeGraaf, R. M., Yamasaki, M., Leak, W. B., & Lester, A. M. (2006). *Technical guide to forest wildlife habitat management in New England*. Burlington, Vermont: University of Vermont Press.

Dettmers, R., & Rosenberg K.V. (2000). *Partners In Flight landbird conservation plan: physiographic area 9: Southern New England.* Retrieved from http://www.partnersinflight.org/bcps/plan/pl_09_10.pdf.

Doherty, P. F., & Grubb, T. C. (2002a). Survivorship of Permanent-Resident Birds in a Fragmented Forested Landscape. *Ecology*, 83(3), 844–857. doi:10.1890/0012-9658(2002)083[0844:SOPRBI]2.0.CO;2

Doherty, P. F., & Grubb, T. C. (2002b). Nest usurpation is an “edge effect” for Carolina chickadees Poecile carolinensis. *Journal of Avian Biology*, 33(1), 77–82. doi:10.1034/j.1600-048X.2002.330112.x

Donnelly, R., & Marzluff, J. (2006). Relative importance of habitat quantity, structure, and spatial pattern to birds in urbanizing environments. *Urban Ecosystems*, 9(2), 99–117. doi:10.1007/s11252-006-7904-2

Donovan, T. M., Jones, P. W., Annand, E. M., & Thompson, F. R. (1997). Variation in Local-Scale Edge Effects: Mechanisms and Landscape Context. *Ecology*, 78(7), 2064–2075.

Dow, D. D. (1969). Habitat utilization by cardinals in central and peripheral breeding populations. *Canadian Journal of Zoology*, 47(3), 409–417. doi:10.1139/z69-075

Ehrhart, R. L., & Conner, R. N. (1986). Habitat selection by the Northern Cardinal in three eastern Texas forest stands. *The Southwestern Naturalist*, 31(2),191–199.

Evans, M. L., Stutchbury, B. J. M., & Woolfenden, B. E. (2008). Off-territory forays and genetic mating system of the Wood Thrush (Hylocichla mustelina). *The Auk*, 125(1), 67–75.

Evans, M., Gow, E., Roth, R.R., Johnson, M.S., & Underwood, T.J. (2011). Wood Thrush (Hylocichla mustelina). (A. Poole, Ed.). *The Birds of North America Online*. Retrieved from http://bna.birds.cornell.edu/bna/species/246 doi:10.2173/bna.246

Foote, J. R., Mennill, D.J., Ratcliffe, L.M. & Smith, S.M. (2010). Black-capped Chickadee (Poecile atricapillus). (A. Poole, Ed.). *The Birds of North America Online*. Retrieved from http://bna.birds.cornell.edu/bna/species/039 doi:10.2173/bna.39

Galli, A. E., Leck, C. F., & Forman, R. T. T. (1976). Avian Distribution Patterns in Forest Islands of Different Sizes in Central New Jersey. *The Auk*, 93(2), 356–364.

Hadidian, J., Sauer, J., Swarth, C., Handly, P., Droege, S., Williams, C., Huff, J., et al. (1997). A citywide breeding bird survey for Washington, D.C. *Urban Ecosystems*, 1(2), 87–102. doi:10.1023/A:1018563125184

Halkin, S. L., & Linville, S. U. (1999). Northern Cardinal (Cardinalis cardinalis). (A. Poole, Ed). *The Birds of North America Online*. Retrieved from http://bns.birds.cornell.edu/bna/species/440 doi:10.2173/bna.440

Holmes, D. (2002). Habitat evaluation procedures (HEP) report; Pend Oreille Wetlands Wildlife II project. Technical Report, Project No. 199106000. Retrieved from https://pisces.bpa.gov/release/documents/documentviewer.aspx?doc=00004008-2

Hoover, J. P., & Brittingham, M. C. (1998). Nest-Site Selection and Nesting Success of Wood Thrushes. *The Wilson Bulletin*, 110(3), 375–383.

Ickes, R. (1992). Northern Oriole, Icterus galbula. In Brauning, D.W. (Ed.), *Atlas of the breeding birds in Pennsylvania* (pp. 410–411). Pittsburgh, PA: University of Pittsburgh Press.

Jackson, J. A. (1976). A Comparison of Some Aspects of the Breeding Ecology of Red-Headed and Red-Bellied Woodpeckers in Kansas. *The Condor*, 78(1), 67–76. doi:10.2307/1366917

Keller, C. M. E., Robbins, C. S., & Hatfield, J. S. (1993). Avian communities in riparian forests of different widths in Maryland and Delaware. *Wetlands*, 13(2), 137–144.

Kessel, B. (1957). A Study of the Breeding Biology of the European Starling (Sturnus vulgaris L.) in North America. *American Midland Naturalist*, 58(2), 257–331. doi:10.2307/2422615

Kilgo, J. C., Sargent, R. A., Chapman, B. R., & Miller, K. V. (1998). Effect of stand width and adjacent habitat on breeding bird communities in bottomland hardwoods. *The Journal of wildlife management*, 62(1), 72–83.

Loery, G., & Nichols, J. D. (1985). Dynamics of a Black-capped Chickadee Population, 1958-1983. *Ecology*, 66(4), 1195–1203. doi:10.2307/1939172

Morse, D. H. (1970). Ecological Aspects of Some Mixed-Species Foraging Flocks of Birds. *Ecological Monographs*, 40(1), 119–168. doi:10.2307/1942443

Mostrom, A. M., Curry, R. L., & Lohr, B. (2002). Carolina Chickadee (Poecile carolinensis). (A. Poole, Ed). *The Birds of North America Online*. Retrieved from http://bna.birds.cornell.edu/bna/species/636 doi:10.2173/bna.636

Mowbray, T. B. (1999). Scarlet Tanager (Piranga olivacea). (A. Poole, Ed.). *The Birds of North America Online*. Retrieved from http://bna.birds.cornell.edu/bna/species/479 doi:10.2173/bna.479

Newton, I. (2004). The recent declines of farmland bird populations in Britain: an appraisal of causal factors and conservation actions. *Ibis*, 146(4), 579–600. doi:10.1111/j.1474-919X.2004.00375.x

Palmer-Ball Jr, B. L. (1996). *The Kentucky breeding bird atlas*. University Press of Kentucky.

Perkins, M. W., Ron J. Johnson, & Blankenship, E. E. (2003). Response of Riparian Avifauna to Percentage and Pattern of Woody Cover in an Agricultural Landscape. *Wildlife Society Bulletin*, 31(3), 642–660.

Pitts, T. D. (1984). Description of American Robin territories in northwest Tennessee. *Migrant*, 55, 1–6.

Price, J., Droege, S., Price, A., & Beadle, D. D. (1995). *The summer atlas of North American birds*. Academic Press San Diego.

Ringler, R. F. (1996). Carolina Chickadee. In C. S. Robbins (Ed.), *Atlas of the Breeding Birds of Maryland and the District of Columbia* (pp. 256–257). University of Pittsburgh Press.

Rising, J.D. & Flood, N. J. (1998). Baltimore Oriole (Icterus galbula), (A. Poole, Ed.). *The Birds of North America Online.* Retrieved from http://bna.birds.cornell.edu/bna/species/384 doi:10.2173/bna.384

Rivera, J. H. V., McShea, W. J., & Rappole, J. H. (2003). Comparison of breeding and postbreeding movements and habitat requirements for the Scarlet Tanager (Piranga olivacea) in Virginia. *Auk*, 120(3), 632–644.

Robbins, C. S. (1980). Effect of forest fragmentation on breeding bird populations in the piedmont of the mid-Atlantic region. *Atlantic Naturalist*, 33(1), 31–36.

Robbins, C. S., Dawson, D. K., & Dowell, B. A. (1989). Habitat area requirements of breeding forest birds of the middle Atlantic states. *Wildlife Monographs*, 3–34.

Roberts, C., & Norment, C. J. (1999). Effects of plot size and habitat characteristics on breeding success of Scarlet Tanagers. *The Auk*, 116(1), 73–82.

Robinson, S. K., Thompson III, F. R., Donovan, T. M., Whitehead, D. R., & Faaborg, J. (1995). Regional forest fragmentation and the nesting success of migratory birds. *Science*, 267(5206),1987–1987.

Roth, R. R., Johnson, M. S., & Underwood, T. J. (1996). Wood Thrush (Hylocichla mustelina). (A. Poole, Ed.). *The Birds of North America Online*. Retrieved from http://bna.birds.cornell.edu/bna/species/246 doi:10.2173/bna.246

Sallabanks, R. & James, F.C. (1999). American Robin (Turdus migratorius), *The Birds of North America Online*. (A. Poole, Ed.). Retrieved from http://bna.birds.cornell.edu/bna/species/462 doi:10.2173/bna.462

Salt, W. R., & Salt, J. R. (1976). *The birds of Alberta: with their ranges in Saskatchewan & Manitoba*. Hurtig.

Schroeder, R.L. (1983). *Habitat suitability index models: Black-capped chickadee.* U.S. Department of the Interior, Division of Biological Services, Research and Development, Fish and Wildlife Service, Washington, DC. Retrieved from http://www.nwrc.usgs.gov/wdb/pub/hsi/hsi-037.pdf

Sedgwick, J. A., & Knopf, F. L. (1990). Habitat Relationships and Nest Site Characteristics of Cavity-Nesting Birds in Cottonwood Floodplains. *The Journal of Wildlife Management*, 54(1), 112–124. doi:10.2307/3808910

Shackelford, C. E., Brown, R. E., & Conner, R. N. (2000). Red-bellied Woodpecker (Melanerpes carolinus). (A. Poole, Ed.). *The Birds of North America Online*. (Retrieved from http://bna.birds.cornell.edu/bna/species/500 doi:10.2173/bna.500

Sharp, M. H. (1990). America's songbird-species profile: American Robin (Turdus migratorius). *Wildbird*, 4, 22–27.

Shy, E. (1984). Habitat shift and geographical variation in North American tanagers (Thraupinae: Piranga). *Oecologia*, 63(2), 281–285.

Smith, S. M. (1991). *The Black-capped Chickadee: behavioral ecology and natural history*. Comstock Pub Assoc.

Smith, S. M. (1993). Black-capped Chickadee (Poecile atricapillus). (A. Poole, Ed.). *The Birds of North America Online*. Retrieved from http://bna.birds.cornell.edu/bna/species/039 doi:10.2173/bna.39

Straus, M. A., Bavrlic, K., Nol, E., Burke, D. M., & Elliott, K. A. (2011). Reproductive success of cavity-nesting birds in partially harvested woodlots. *Canadian Journal of Forest Research*, 41(5), 1004–1017. doi:10.1139/X11-012

Sturman, W. A. (1968). Description and analysis of breeding habitats of the chickadees, Parus atricapillus and P. rufescens. *Ecology*, 49(3), 418–431.

Thompson III, F. R., Robinson, S. K., Donovan, T. M., Faaborg, J. R., Whitehead, D. R., & Larsen, D. R. (2000). Biogeographic, landscape, and local factors affecting cowbîrd abundance and host parasitism levels. In Smith, J. N. M., Cook, T. L., Rothstein, S. I., Robinson, S. K., Sealy, S. G. (Eds.), *Ecology and Management of Cowbirds and Their Hosts: Studies in the Conservation of North American Passerine Birds* (pp. 271-279). Austin, TX- University of Texas.

Tirpak, J. M., Jones-Farrand, D. T., Thompson III, F. R., Twedt, D. J., & Uihlein III, W. B. (2009). *Multiscale habitat suitability index models for priority landbirds in the Central Hardwoods and West Gulf Coastal Plain/Ouachitas Bird Conservation Regions* (General Technical Report NRS-49). Newtown Square, PA, U.S. Department of Agriculture, Forest Service, Northern Research Station, USA. Retrieved from http://www.nrs.fs.fed.us/pubs/9723i

Tilghman, N. G. (1987). Characteristics of urban woodlands affecting breeding bird diversity and abundance. *Landscape and Urban Planning*, 14, 481–495.

Twomey, A. C. (1945). The bird population of an elm-maple forest with special reference to aspection, territorialism, and coactions. *Ecological Monographs*, 15(2), 173–205.

Watts, B.D. (1999). *Partners in Flight: Mid-Atlantic Coastal Plain bird conservation plan: physiographic Area #44*. Retrieved from http://www.partnersinflight.org/bcps/plan/pl_44_10.pdf.

Weinberg, H. J., & Roth, R. R. (1998). Forest area and habitat quality for nesting Wood Thrushes. *The Auk*, 115(4),879–889.

Winkler, H., Christie, D. A., & Nurney, D. (1995). Woodpeckers: an identification guide to the woodpeckers of the world. *Houghton Mifflin Company, New York*.

Young, H. F. (1950). *Territorial behavior in the eastern robin*. University of Wisconsin–Madison.

Zumeta, D. C., & Holmes, R. T. (1978). Habitat shift and roadside mortality of Scarlet Tanagers during a cold wet New England spring. *The Wilson Bulletin*, 90(4), 575–586.

Table 1. Relationship between canopy percent per 0.04 ha and suitability index (SI) for American robin (AMRO) habitat, and associated references.

Canopy Percent (per 0.04 ha)	SI Score (AMRO)	Reference
0	0.7	assumed value
10	0.8	assumed value
20	0.9	assumed value
25	1	assumed value
30	1	assumed value
35	1	assumed value
40	0.6	assumed value
50	0.4	assumed value
70	0.3	assumed value
90	0.2	assumed value
100	0.1	assumed value

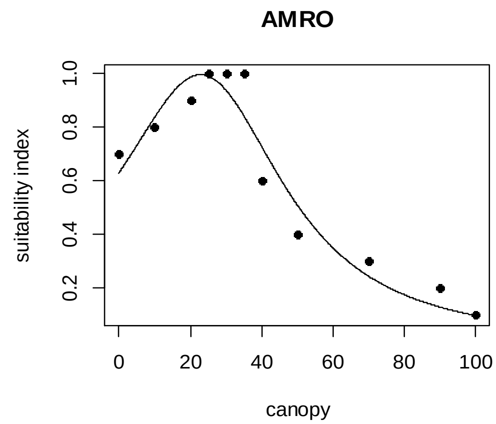


Table 2. Relationship between lawn percent per 0.04 ha and suitability index (SI) for American robin (AMRO) habitat, and associated references.

Lawn Percent (per 0.04 ha)	SI Score (AMRO)	Reference
0	0.2	assumed value
40	0.6	assumed value
80	1	assumed value
100	0.8	assumed value

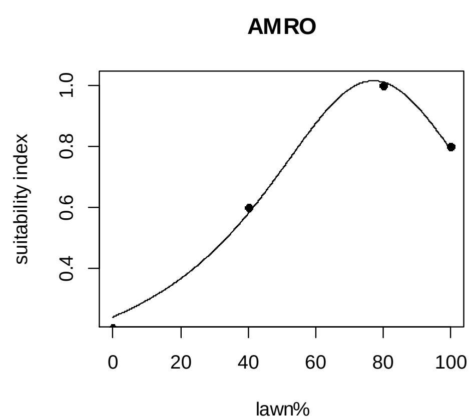


Table 3. Relationship between canopy percent per 0.04 ha and suitability index (SI) for Baltimore oriole (BAOR) habitat, and associated references.

Canopy Percent (per 0.04 ha)	SI Score (BAOR)	Reference
0	0	assumed value
10	0.3	assumed value
20	0.8	assumed value
30	1	DeGraaf et al., 2006
50	1	assumed value
70	0.2	Gotfryd, 1984
80	0.1	Palmer-Ball, 1996
100	0	assumed value

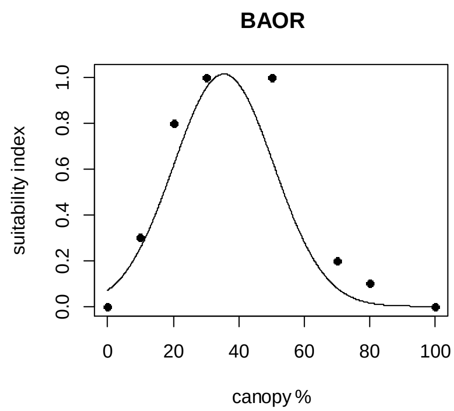


Table 4. Relationship between large tree density (dbh >23cm) and suitability index (SI) for Baltimore oriole (BAOR) habitat, and associated references.

Large Tree Density (per 0.04 ha)	SI Score (BAOR)	Reference
0	0	Perkins et al., 2003
1	0.5	assumed value
3	1	DeGraaf et al., 2006
5	0.7	assumed value
7	0.5	assumed value
9	0.3	assumed value
11	0.1	assumed value

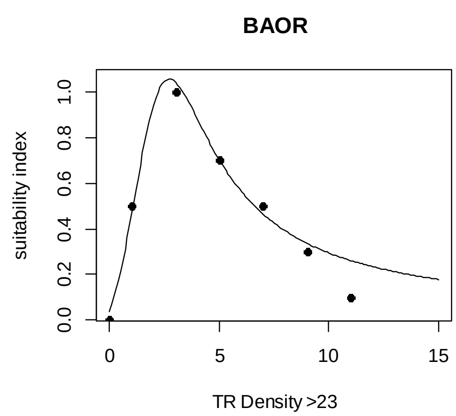


Table 5. Relationship between canopy percent per 0.04 ha and suitability index (SI) for black-capped chickadee (BCCH) habitat, and associated references. Carolina chickadee habitat models mimic black-capped chickadee.

Canopy Percent (per 0.04 ha)	SI Score (BCCH)	Reference
0	0	Holmes, 2002
12	0.25	Holmes, 2002
37	0.7	Holmes, 2002
50	1	Holmes, 2002
75	1	Holmes, 2002
87	0.8	Holmes, 2002

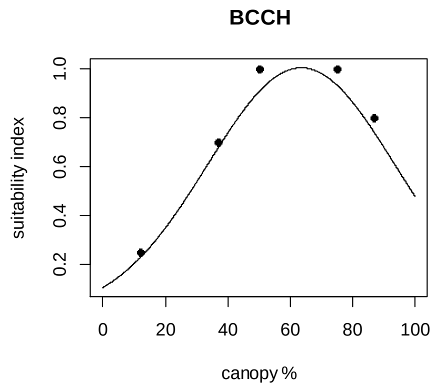


Table 6. Relationship between tree height (m) and suitability index (SI) for black-capped chickadee (BCCH) habitat, and associated references.

Mean Tree Height (m) (per 0.04 ha)	SI score (BCCH)	Reference
0	0	Holmes, 2002
1	0.15	Holmes, 2002
5	0.5	Holmes, 2002
10	0.85	Holmes, 2002
15	1	Holmes, 2002

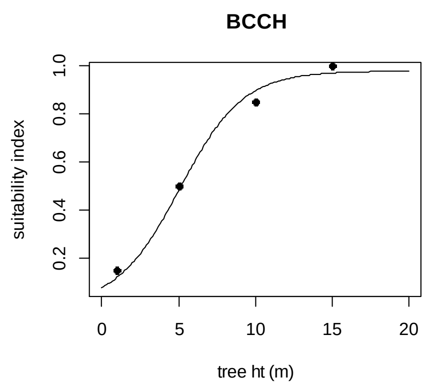


Table 7. Relationship between dead wood density per ha and suitability index (SI) for black-capped chickadee (BCCH) habitat, and associated references.

Dead Wood Density (per 0.04 ha)	SI Score (BCCH)	Reference
0	0.03	Holmes, 2002
4	0.9	Holmes, 2002
6	1	Sedgwick & Knopf, 1990

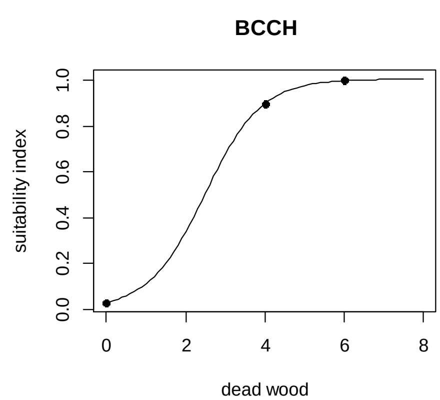


Table 8. Relationship between lawn percent per 0.04 ha and suitability index (SI) for European starling (EUST) habitat, and associated references.

Lawn Percent (per 0.04 ha)	SI Score (EUST)	Reference
0	0	assumed value
20	0.2	assumed value
40	0.6	assumed value
60	1	assumed value
80	1	assumed value
100	1	assumed value

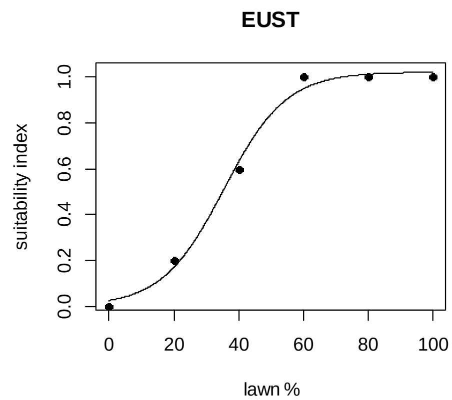


Table 9. Relationship between percent building and suitability index (SI) for European starling (EUST) habitat, and associated references.

Building Percent (per 0.04 ha)	SI Score (EUST)	Reference
0	0	assumed value
25	0.8	assumed value
50	1	assumed value
75	0.6	assumed value
100	0.4	assumed value

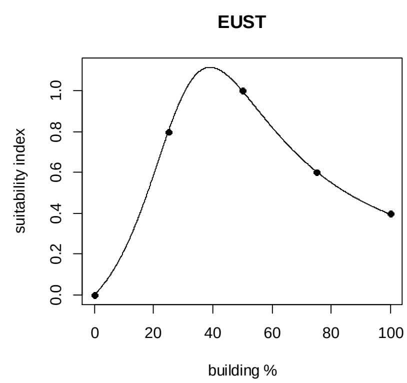


Table 10. Relationship between total tree density per 0.04 ha and suitability index (SI) for European starling (EUST) habitat, and associated references.

Tree Density (per 0.04 ha)	SI Score (EUST)	Reference
0	0.8	assumed value
1	1	assumed value
3	1	assumed value
5	0.5	assumed value
7	0.1	assumed value
10	0	assumed value

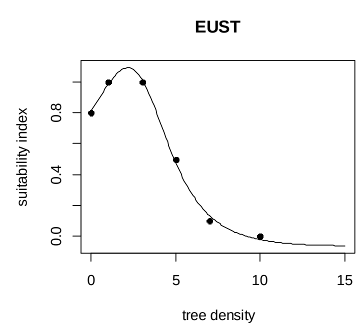


Table 11. Relationship between dead wood density per ha and suitability index (SI) for European starling (EUST) habitat, and associated references.

Dead Wood Density (per 0.04 ha)	SI Score (EUST)	Reference
0	0.06	Strauss et al., 2011
1	0.93	Strauss et al., 2011
3	1	Adkins et al., 2003

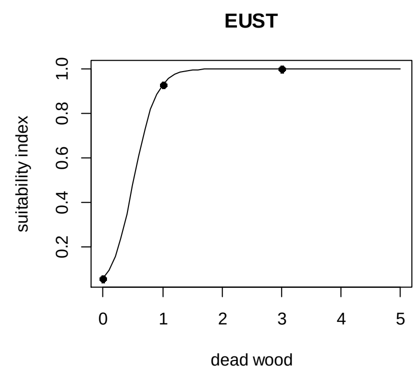


Table 12. Relationship between canopy percent per 0.04 ha and suitability index (SI) for northern cardinal (NOCA) habitat, and associated references.

Canopy Percent (per 0.04 ha)	SI Score (NOCA)	Reference
0	0.7	assumed value
10	0.8	assumed value
20	0.9	assumed value
27	1	Conner et al., 1986
30	1	assumed value
32	1	assumed value
40	0.6	assumed value
50	0.4	assumed value
70	0.2	assumed value
90	0.1	assumed value
100	0	assumed value

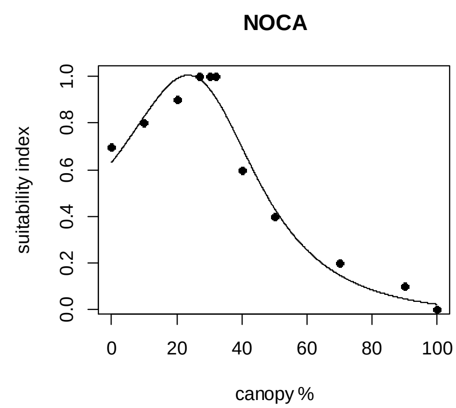


Table 13. Relationship between shrub percent and suitability index (SI) for northern cardinal (NOCA) habitat, and associated references.

Shrub Percent (per 0.04 ha)	SI Score (NOCA)	Reference
0	0	assumed values
20	0.6	assumed values
40	0.8	assumed values
60	1	assumed values
80	1	assumed values
100	0.8	assumed values

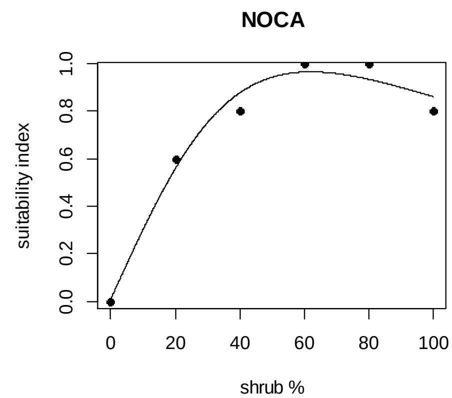


Table 14. Relationship between large tree density (trees larger than 23 cm dbh) per 0.04 ha and suitability index (SI) for red-bellied woodpecker (RBWO) habitat, and associated references.

Large Tree density (per 0.04 ha)	SI score (RBWO)	Reference
0	0	Assumed value
3	0.6	Assumed value
6	1	Adkins et al., 2003
8	0.9	Conner, 1980
11	0.8	Assumed value

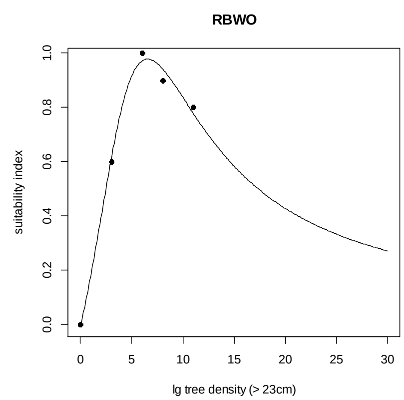


Table 15. Relationship between basal area (trees > 6 cm dbh) per ha and suitability index (SI) for red-bellied woodpecker (RBWO) habitat, and associated references.

Basal Area (per ha)	SI Score (RBWO)	Reference
0	0	Assumed value
4	0.5	Assumed value
8	0.95	Conner, 1980 (based on SD)
14	1	Conner, 1980
34	1	Adkins et al., 2003

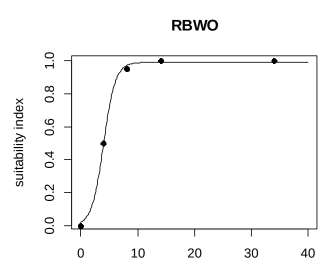


Table 16. Relationship between canopy percent per 0.04 ha and suitability index (SI) for red-bellied woodpecker (RBWO) habitat, and associated references.

Canopy Percent (per 0.04 ha)	SI Score (RBWO)	Reference
0	0	assumed value
15	0.1	assumed value
20	0.3	assumed value
25	0.5	assumed value
35	0.9	DeGraaf et al., 2006
62	1	Strauss et al., 2011

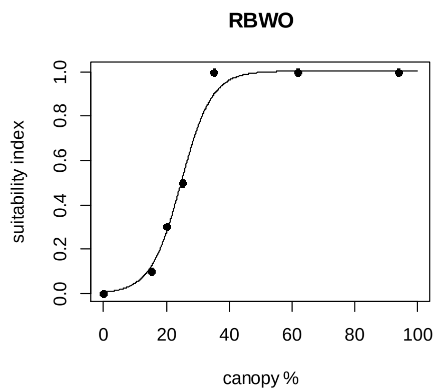


Table 17. Relationship between dead wood density per ha and suitability index (SI) for red-bellied woodpecker (RBWO) habitat, and associated references.

Dead Wood Density (per 0.04 ha)	SI Score (RBWO)	Reference
0	0.06	Strauss et al., 2011
1	0.93	Strauss et al., 2011
3	1	Adkins et al., 2003

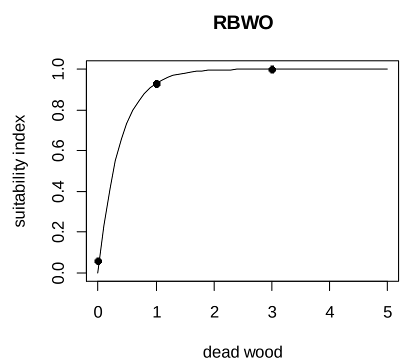


Table 18. Relationship between large tree density (trees larger than 23 cm dbh per 0.04 ha) and suitability index (SI) for scarlet tanager (SCTA) habitat, and associated references.

Large Tree Density (per 0.04 ha)	SI Score (SCTA)	Reference
0	0	assumed value
10	0.1	assumed value
15	0.4	assumed value
20	1	Roberts & Norment, 1999
26.2	1	Shy, 1984
32	1	Rivera et al., 2003

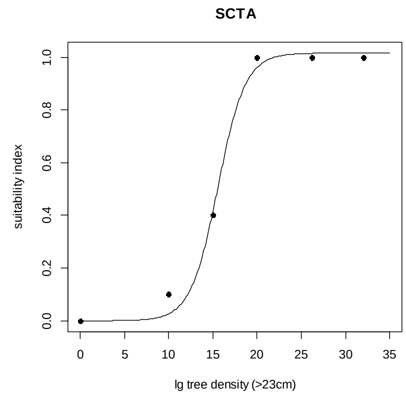


Table 19. Relationship between basal area (trees >6 cm dbh) per ha and suitability index (SI) for scarlet tanager (SCTA) habitat, and associated references.

Basal Area (per ha)	SI Score (SCTA)	Reference
0	0	Assumed value
10	0.05	Assumed value
20	0.2	Assumed value
30	0.3	Assumed value
40	0.7	Assumed value
50	0.8	Assumed value
62	1	Roberts & Norment, 1999

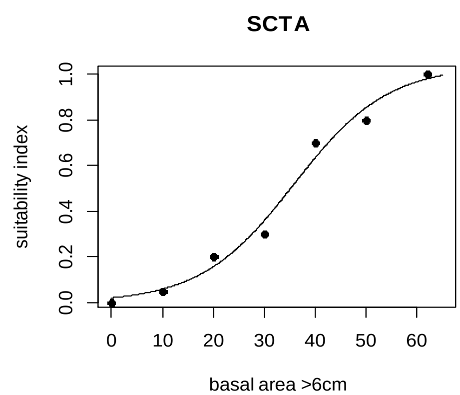


Table 20. Relationship between canopy percent per 0.04 ha and suitability index (SI) for scarlet tanager (SCTA) habitat, and associated references.

Canopy Percent (per 0.04 ha)	SI Score (SCTA)	Reference
0	0	assumed value
50	0.2	assumed value
75	0.95	Shy, 1984
89	1	Shy 1984, Roberts and Norment, 1999

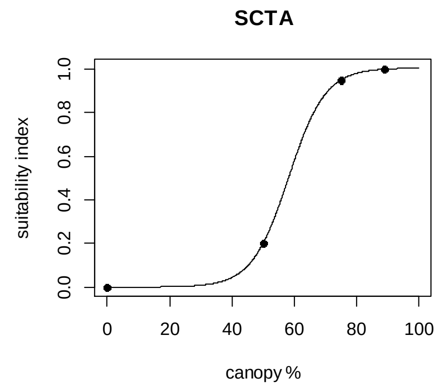


Table 21. Relationship between contiguous forest area (ha) surrounding the i-Tree plot and suitability index (SI) for scarlet tanager (SCTA) habitat, and associated references.

Forest Area (ha)	SI Score (SCTA)	Reference
0	0	assumed value
1	0.3	Robbins et al., 1989
10	0.5	Robbins et al., 1989
32	0.6	Robbins et al., 1989
100	0.7	Robbins et al., 1989
300	0.8	Robbins et al., 1989
500	0.9	Robbins et al., 1989

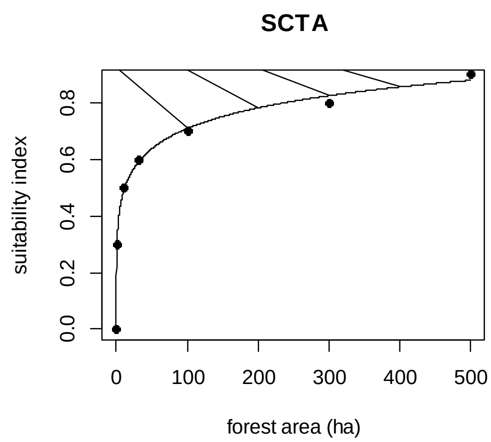


Table 22. Relationship between canopy percent per 0.04 ha and suitability index (SI) for wood thrush (WOTH) habitat, and associated references.

Canopy Percent (per 0.04 ha)	SI Score (WOTH)	Reference
25	0	Hoover & Brittingham, 1998
70	0.25	Annand & Thompson, 1997
90	0.9	Annand & Thompson, 1997
100	1	Annand & Thompson, 1997

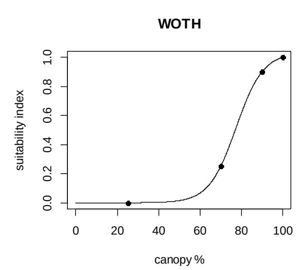


Table 23. Relationship between sapling density and suitability index (SI) for wood thrush (WOTH) habitat, and associated references.

Sapling Density (per 0.04 ha)	SI Score (WOTH)	Reference
0	0	assumed value
2	0.1	assumed value
5	0.4	assumed value
7	0.8	assumed value
10	1	assumed value

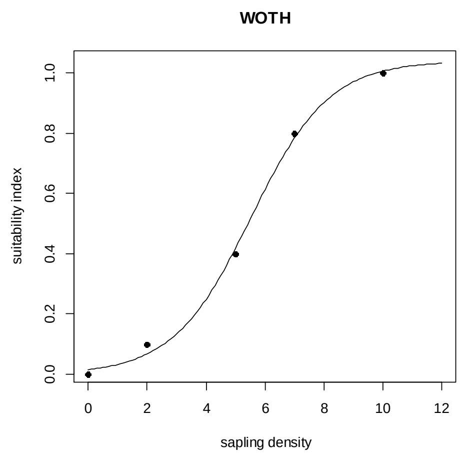


Table 24. Relationship between percent forest within a1km radius of the i-Tree plot and suitability index (SI) for wood thrush (WOTH) habitat, and associated references.

% Forest (1km)	SI Score (WOTH)	Reference
0	0	assumed value
10	0	assumed value
20	0.05	assumed value
30	0.1	Donovan et al., 1997
40	0.25	assumed value
50	0.5	Donovan, et al., 1997
60	0.75	assumed value
70	0.9	Donovan et al., 1997
80	0.95	assumed value
90	1	Donovan et al., 1997
100	1	assumed value