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Research Paper

# 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 |

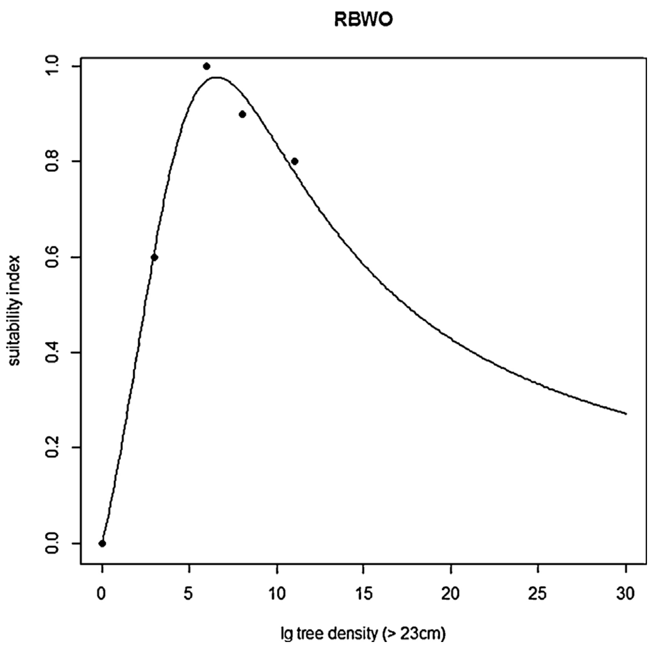


**Fig. 1.** Relationship between large tree density (trees larger than 23 cm dbh) per 0.04 ha and suitability index (SI) for red-bellied woodpecker (RBWO) habitat, and associated references.

**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) |

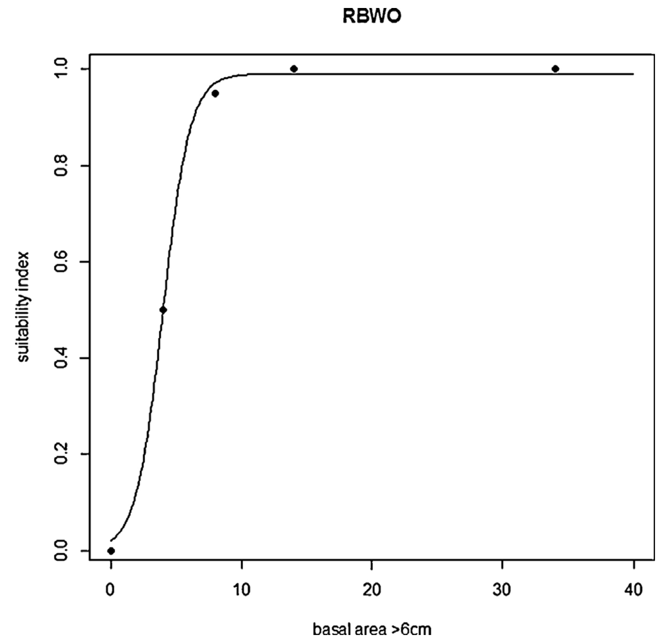


**Fig. 2.** Relationship between basal area (trees > 6 cm dbh) per ha and suitability index (SI) for red-bellied woodpecker (RBWO) habitat, and associated references.

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 $1/(1+(15.67 \times \exp(-5.338 \times \text{dead wood density per 0.04 ha})))$ (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.

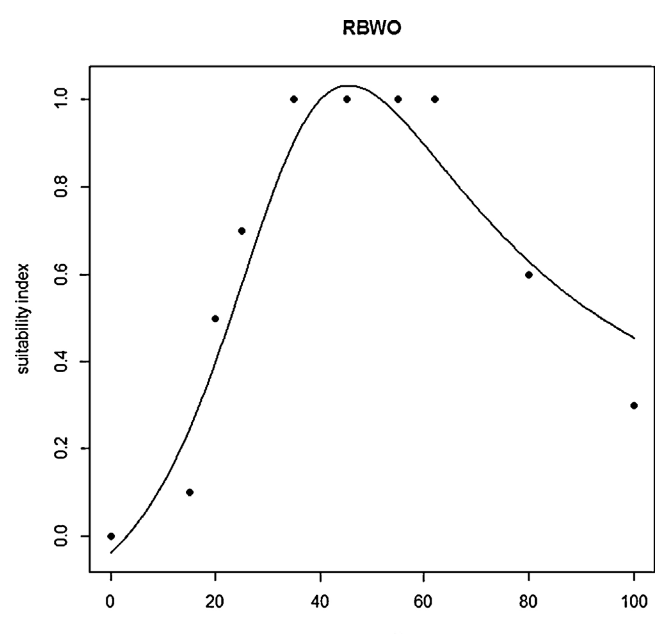


**Fig. 3.** Relationship between canopy percent per 0.004 ha and suitability index (SI) for red-bellied woodpecker (RBWO) habitat, and associated references.

**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 ($P = 0.0140$, $r_s = 0.2705$).

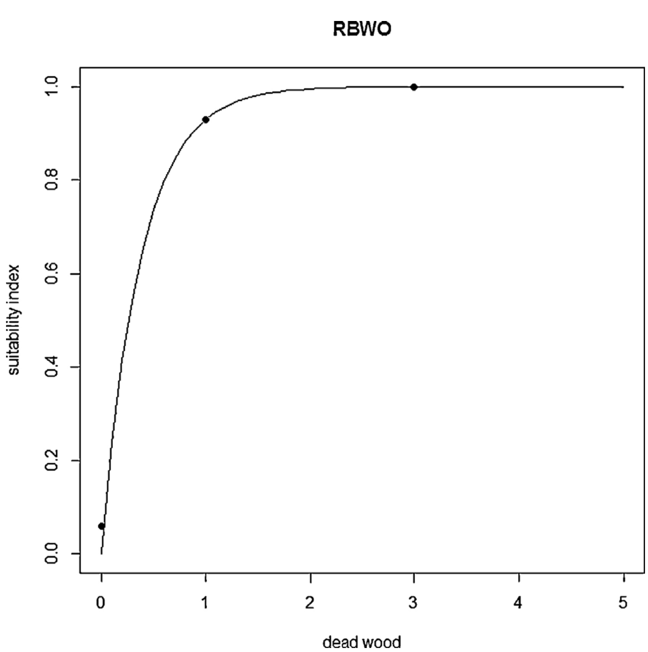


**Fig. 4.** Relationship between deadwood density per ha and suitability index (SI) for red-bellied woodpecker (RBWO) habitat, and associated references.

### 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.

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