



# Native plants improve breeding and foraging habitat for an insectivorous bird



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## ABSTRACT

Understanding how introduced plants reduce food web complexity is critical to effective conservation management within human-dominated systems. In urban breeding birds, the paucity of dietary specialists suggests that a lack of food resources, such as arthropod prey essential for reproduction and survival, may contribute to bird declines. Local plant species composition and abundance is influenced by the landscaping decisions of private homeowners and may be contributing to differences in insect prey availability. In this study, we examined whether non-native plants are a limiting factor to a resident breeding insectivore, the Carolina chickadee (*Poecile carolinensis*). We used caterpillar counts, chickadee foraging observations and detection-corrected hierarchical models, to determine the influence of local landscaping features on insect food availability, chickadee tree preference, site occupancy, site abundance and breeding territory selection. Native plants were more likely to host a higher biomass of caterpillars compared to non-native plants, and chickadees strongly preferred to forage in native plants that supported the most caterpillars. In addition, chickadees were less likely to breed in yards as the dominance of non-native plants increased. Chickadee occupancy increased with tree basal area and chickadee abundance declined as impermeable surface area increased and basal area decreased. Our results demonstrate that non-native plants reduce habitat suitability for chickadees by reducing insect food available for breeding. Improving human-dominated landscapes as wildlife habitat should include increasing native, and arthropod-producing, plant species to effectively support the life history needs of insectivorous birds.

## 1. Introduction

Worldwide, habitat is rapidly being converted from coevolved native ecosystems into novel assemblages of plants and animals (Radeloff et al., 2015). Nowhere are these changes more apparent than within the human-dominated residential matrix. Urban-associated declines in the abundance and richness of native organisms have been documented globally (Dolan et al., 2011; McKinney, 2008). Because conversion to ‘urban’ development includes a variety of concurrent changes to the local ecosystem, conservation ecologists have called for a mechanistic understanding of the drivers underlying species declines in these systems (Shochat et al., 2006).

One of the most ubiquitous threats to biodiversity today is the conversion of native plant communities into plant assemblages dominated by non-native species (Johnson, 2007). Such conversions have triggered debate about the benefit of managing non-native species particularly when it is unclear how well introduced plants support wildlife and management is financially and logistically challenging. From a conservation perspective, this debate cannot be resolved

without a clear understanding of both the positive and negative impacts of non-native plants. Unfortunately, there are few studies that have examined whether introduced plants provide ecological niches that are equivalent to the native species that are displaced (Tallamy, 2004). Needed are multi-trophic studies of native and non-native plants that elucidate how differences in bottom-up resources affect higher-order consumers in novel ecosystems (Faeth et al., 2005; Harvey et al., 2010).

Recent studies suggest that, on average, consumer biodiversity, particularly the abundance, richness and survival of herbivorous insects, is reduced by non-native plants (Burghardt et al., 2010; Holmquist et al., 2011; Litt and Steidl, 2010; Tallamy et al., 2010). This occurs in part because herbivorous insects have adapted to circumvent the phytochemical defenses of particular plant lineages, resulting in a radiation of specialized plant-insect associations (Forister et al., 2015). During urban conversion, native plants are replaced by non-native species with novel chemical, physical, and phenological features for which native herbivorous arthropods have few physiological or behavioral adaptations. This can result in reduced herbivory on introduced plants and a competitive advantage for these plants to spread (i.e.

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Enemy Release Hypothesis; Keane and Crawley, 2002).

It is well documented that the biomass of arthropods, particularly Lepidoptera larvae, supports large and diverse trophic webs, and is an important component of the diets of insectivorous consumers such as birds (Cooper, 1988; Holmes and Schultz, 1988). Even generalist bird species rely heavily on arthropods during the breeding season because these food items provide high protein, calcium, and carotenoids for nestling growth (Eeva et al., 2010; Razeng and Watson, 2014). Thus, landscaping with non-native plants may negatively affect bird populations if individuals preferentially rely on herbivorous insects and non-native plants do not support adequate prey populations for breeding birds. In contrast, non-native plants could promote increases in other food items (e.g. non-native arthropods), keeping overall prey biomass similar between native and non-native plants (Cook and Talley, 2014; Mitchell and Litt, 2016) and bird populations unaffected. Exploring the trajectory of these relationships requires simultaneous study of insect communities and bird populations in the presence of both native and non-native plants.

Plant abundance and species composition in residential areas are primarily a result of landscaping decisions of homeowners and developers on private land (Lerman and Warren, 2011). Interest in ‘rewilding suburbia’ has sparked renewed public attention for landscaping that contributes to wildlife habitat (Marzluff, 2014; Tallamy, 2007). For example, population expansion of the rare *Eumaeus atala* butterfly resulted from increases in the horticultural sale of native *Zamia* sp., the sole genus of host plant for this species (Culbert, 2013). If local factors that drive population persistence within a residential patch are identified, this information could assist landowners in providing additional resources for wildlife, and help increase native biodiversity in these systems (Goddard et al., 2010).

In this study, we used the Carolina chickadee (*Poecile carolinensis*; hereafter, ‘chickadee’) as a representative insectivorous bird to investigate how plant species origin influences foraging and breeding behavior in residential neighborhoods. Specifically, we followed foraging behaviors of individually marked birds to determine if chickadees exhibit a preference for native over non-native plants. In addition, we used hierarchical models to determine which local habitat features predict occupancy, abundance and nesting activity of chickadees. Given their insectivorous diet during the breeding season, we tested the hypothesis that both plant species origin (native or non-native) and consumer productivity (i.e. the probability of supporting Lepidoptera prey) influences the occurrence of chickadees as well as their foraging and breeding decisions. We predicted that areas with more native plants would support more chickadees, and chickadees would forage more often in the most insect-producing native plants.

## 2. Methods

### 2.1. Study species

Chickadees are year-round residents that inhabit Eastern deciduous forests as well as residential areas. During the breeding season (this region: April–early June) arthropod prey make up > 90% of chickadee diet, particularly Lepidoptera larvae, Hemiptera, and Araneae (Mostrom et al., 2002). Chickadees are single-brooded, synchronous, cavity nesters that readily use artificial nest boxes.

### 2.2. Study sites

Our study took place between March–June in 2013–2014 within private residential yards of homeowners who volunteered for the Smithsonian’s Neighborhood Nestwatch program in the Washington D.C. metropolitan area (Evans et al., 2005, Yard Locations: Supplementary Fig. 1). We selected 97 sites from a pool of 195 yards; most were separated by at least 1 km (Mean distance:  $22.26 \pm 0.16$  km). Inclusion in this study was primarily driven by access permission;

however, sites were distributed across a rural-urban landscape gradient and in areas of varying human population density and socioeconomic status (Lerman and Warren, 2011). Prior to data collection, all sites received an artificial cavity nesting tube (modified from Grubb and Bronson, 1995) to assure that site occupancy would not be influenced by the availability of suitable nesting locations. Although our nest box and point count sampling took place within the focal yard, we aimed to conduct our plant, caterpillar and chickadee behavior at a larger, patch scale that was relevant to the size of a chickadee territory (Goddard et al., 2010). Thus, these samples took place within a 50-m radius surrounding the focal yard which included neighboring properties (Supplementary Fig. 2). Caterpillar and chickadee foraging data was collected in both 2013 and 2014 and plant communities. Chickadee occupancy, abundance and chickadee breeding data were collected in 2014.

### 2.3. Caterpillar sampling

To determine the caterpillar abundance on individual woody plant species, we conducted a timed-search sampling effort, designed for detection of Lepidoptera on woody plants, within a subset of yards where chickadees were present (Wagner, 2010; Burghardt et al., 2010). Sampling was conducted between May and early June to encompass the period when chickadees were feeding young, and to only sample one peak of caterpillar biomass. Plants were selected by walking 25 m from the center of the yard in each cardinal direction and sampling the four plant species encountered (total: 16 plants per site). For 5 min the observer meticulously searched foliage and stems counting and collecting all folivorous holometabolous larvae (mostly Lepidoptera but also Hymenoptera sawflies; hereafter ‘Caterpillars’) located in an area on the plant (approximately 1.5 m × 1.5 m) up to 4 m high, and measured each caterpillar to the nearest 0.5 mm. Each five-minute search period was repeated three times per plant species per site on different areas of vegetation (total: 48 five-minute samples per yard).

### 2.4. Foraging behavior

Adult chickadees breeding at the site were captured to attach unique color band combinations for re-identification. To quantify foraging effort on plants, observers systematically surveyed the focal yard, accessible neighboring yards, and adjacent public land to record foraging behavior of the breeding pair. Once a color-banded bird was located, plant species used for foraging were recorded every minute (2014) or every plant switch (2013) until the bird was lost; observations resumed when the focal individual was relocated. We confirmed active foraging by observing searching and/or probing behavior, and the absence of other non-feeding behavior (i.e. singing, preening, etc.). Sites were visited every 2–5 days while the nest was active, alternating observers, and observations were attempted for a minimum of 1 h per visit.

### 2.5. Bird surveys

Surveys were conducted from 15 Apr–14 Aug 2014. We surveyed each site 2–3 times and all surveys were completed in the morning between 0630 and 1100 when bird activity is highest. During a 10-minute observation period, a trained observer identified all chickadees that were seen or heard within a 50 m radius. The central point of the survey was located approximately 10 m from the backside of the house in a location that maximized coverage of the focal residential yard. For occupancy analyses, we pooled abundance per survey into a binary response so that chickadees were either detected (= 1) or not detected (= 0) at each site per visit. For abundance analyses, we used the maximum number of individuals observed at each site per visit. Because chickadee territories begin to break down and fledglings disperse in June and July in this region (Mostrom et al., 2002), we included only the 1st and 2nd survey visits (i.e. April–May) for these analyses.

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